# TECHNOLOGIES TO MAINTAIN BIOLOGICAL DIVERSITY

## Volume II--Contract Papers

## Part C: Natural Systems - U.S.

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7. Federal Laws and Policies Pertaining to the Maintenance of Biological Diversity on Federal and Private Lands
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8. Ecological Restoration As A Strategy for Conserving Biological Diversity
   *(William R. Jordan, III, Edith B. Allen, and Robert L. Peters, II)*

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Contract Paper No. 1

# Status/Trends of U.S. Natural Ecosystems

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Office of Technology Assessment
U.S. Congress

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# STATUS AND TREND OF NATURAL ECOSYSTEMS IN THE UNITED STATES

A Report to the Office of Technology Assessment (OTA)
of the Congress of the United States

by

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# Abstract

Recognition of different types of natural ecosystems is a necessary prerequisite to programs of ecosystem protection. Natural ecosystems are usually identified and classified by one of two main approaches: (1) taxonomic components, which establishes land units by grouping resources or sites with similar natural properties, and (2) regionalization, which subdivides land into natural units on the basis of spatial patterns that reflect natural processes. Many terrestrial ecosystems can be conveniently described on the basis of their dominant vegetation types. Several methods that provide taxonomic classifications of the vegetative components of ecosystems are discussed. In addition, state natural heritage inventory programs provide taxonomic components classifications for a wide array of vegetated and non-vegetated terrestrial and aquatic ecosystems, and the U.S. Fish and Wildlife Service has devised a similar but much more detailed classification for aquatic ecosystems. Regionalizations that have been proposed or used to classify U.S. terrestrial and aquatic ecosystems are also discussed, including the biogeographical province and ecoregion methods. Integration of the taxonomic components and regionalization approaches can be used to identify natural ecosystem diversity in considerable detail at regional and national levels. Varying amounts of protection for U.S. ecosystem diversity are provided by public and private land designations, the most extensive being those found in Federal lands managed by the National Park Service, Forest Service, U.S. Fish and Wildlife Service, and Bureau of Land Management. Many state and local (county/city) areas also have various designations. Levels of protection range from the highest, e.g. privately owned and legally secured single-purpose nature reserves, to the lowest, e.g. areas protected only through voluntary cooperation of owners or managers, and areas such as Federal military installations and state school lands on which ecosystem conservation is not emphasized and there are other ascendant priorities. Recent, conservative estimates of the proportion of major terrestrial ecosystem types that are not protected in much of the Federal domain vary from 21 to 51%, depending on the size and number of examples of each type thought to be needed for adequate protection. A study conducted at the Oak Ridge National Laboratory estimates that different major natural ecosystem types have decreased in actual or potential area from 89% (Tule Marshes) to 1% (Alpine Meadows and Barren) since presettlement times. Considering all types, the states of Iowa, Illinois, and Indiana were estimated to have lost 82 to 92% of their actual or potential natural ecosystem diversity, while Nevada, Arizona, and New Hampshire have lost only 4 to 12%. Large losses of certain azonal ecosystem types not normally included in regional and U.S. assessments of potential natural diversity have also occurred. For example, the overall decline in major U.S. wetland ecosystem types from presettlement times is 54% according to a recent U.S. Fish and Wildlife Service estimate. From 70 to 90% of the original riparian ecosystems of the United States are believed to have been lost or greatly disturbed, with the most serious impacts having occurred in the arid and semiarid western states. More detailed estimates of actual (existing) ecosystem diversity should eventually be provided by programs such as the state heritage inventories of The Nature Conservancy and individual states, western U.S. habitat type analyses of the U.S. Forest Service and its collaborators, the integrated habitat inventory and classification program of the Bureau of Land Management in the western United States, and the RPA/RCA assessments of the U.S. Forest Service and U.S. Soil Conservation Service. The latter are required by law and have the potential to provide nationwide estimates of changes in terrestrial and aquatic ecosystem diversity every 10 years. Remote sensing techniques currently being developed may also provide useful information on certain types of ecosystem diversity in the future. Continuing efforts to determine status and change in U.S. ecosystem diversity should be supported by public and private means, as they help greatly to prioritize protection programs and make them cost effective. A classification that more accurately describes the actual level of ecosystem protection in different types of designated areas is also needed to make protection programs more effective. Information obtained for this report suggests that an average of hundreds (but not thousands) of terrestrial and aquatic ecosystem types per state may need to be protected in a comprehensive national preservation system. The number and size of additional preservation units required for this purpose are presently unknown and will depend on the results of extensive inventory programs and preserve design considerations.

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## I. Introduction

This report will review and assess the different natural ecosystem classifications used in the United States, the land designations established to maintain ecosystem and biological diversity, the extent to which different ecosystem types are located on lands that provide protection for biological diversity, and the evidence that changes are occurring in U.S. ecosystem diversity. This information will be used for the "maintenance of biological diversity in situ" part of OTA's assessment of "Technologies to Maintain Biological Diversity."

There is a very large amount of information available on natural ecosystem classification and many approaches are currently being used in the United States. This subject will necessarily be treated selectively, with emphasis on application to conservation of diversity. Numerous land designations that provide differing levels of protection to natural ecosystems have been established and used in the public (Federal, state, local) and private sectors. A reasonably complete list of these designations will be provided together with a brief assessment of their effectiveness. Little information is available concerning the extent to which ecosystem diversity is currently protected on designated lands in the United States or on changes that are occurring in diversity. The material that has been obtained on these topics will be discussed in some detail.

## II. Natural Ecosystem Classification

An ecosystem is an area on or near the earth's surface within which there are associations of biotic and abiotic factors (Fosberg, 1967; Bailey, 1985). This may include sites on the order of a few hundred acres, landscape mosaics of several hundred thousand acres, or regions that encompass millions of acres. Natural ecosystems are those that would occur naturally without human intervention, as well as those which have been disturbed by human factors but still retain a reasonable level of naturalness. For example, managed forests and rangelands often fit this description, whereas cultivated fields and urban parks usually do not.

Vegetation classification will be used in this report to refer to classification of identifiable units of vegetation or "plant communities" into various taxonomic categories such as plant associations, series, and formations, often in a hierarchical or nested scheme. This is different from ecosystem classification, which identifies categories based on consideration of several ecosystem components such as vegetation, soils, and landforms. However, since vegetation is the most easily observable manifestation of climate, soils, topography, etc. in many terrestrial ecosystems, it is often used to name and describe ecosystem types. Because faunal distribution patterns tend to correlate with those of vegetation, little attention has been paid to the wildlife component in recent ecosystem classification and mapping endeavors (Bailey, Pfister and Henderson, 1978). This is also partly a result of the difficulty in obtaining information over a short time on organisms with complex behavior.

Aquatic components such as lakes and streams have received much less attention than vegetation from the standpoint of classification and mapping. They are difficult to map at small scale and how they should be integrated into ecological land classification is poorly understood. The same is true of riparian (streamside) vegetation, mountain meadows, and other azonal types for which distributions do not conform well to regional macroclimates. Bailey

(1982) has called for more emphasis on integration of wildlife and aquatic components in ecosystem classification and mapping.

Before discussing some of the classification and mapping methods used on national, regional, state, and local levels, two important approaches to the problem must be distinguished. Taxonomic classification establishes land units by grouping resources or sites with similar properties, whereas regionalization subdivides land into natural units on the basis of spatial patterns that affect resource use and natural processes (Bailey, Pfister and Henderson, 1978). Taxonomic classification usually begins with a large number of basic units and then aggregates them on the basis of similarities into larger and larger hierarchical groups. On the other hand, regionalization may begin with a whole continent and then subdivide it into progressively smaller, more closely related units within a hierarchical system. The term "classification" is used here and later in a general sense. More specifically, it refers to placing objects into abstract groups on the basis of their similarities or relationships. This does not necessarily provide larger or juxtaposed land units. In this stricter usage, regionalization would not be considered a type of classification (for further discussion, see Bailey, Pfister and Henderson, 1978).

There are some strongly held differences of opinion concerning the taxonomic and regionalization classification and mapping approaches that pertain directly to programs for maintaining biological diversity. A satisfactory method of identifying, describing, and classifying the major elements or components of biological diversity (plants, animals, terrestrial communities, and aquatic communities) is obviously basic to an efficient program of conserving these components. The taxonomic or "components" approach has certain important advantages for this purpose (Radford, 1978; U.S. Department of the Interior, Heritage Conservation and Recreation Service, 1978). Each component is identified and classified. For example, separate taxonomies are used to identify flora, fauna, and soils. Individual occurrences of high priority components can then be protected by establishing natural areas that contain them. This avoids a problem inherent in the regionalization approach, viz., the difficulty in identifying a spatial unit that adequately and jointly represents distributions of the multiple components on which the regionalization scheme is based. Thus the single boundary between regions will usually be a compromise between, say, the distributions of flora, soils, and landforms.

A major advantage of regionalization is that it displays spatial patterns and allows inferences about ecosystem processes that would be difficult to deduce from component level observations alone (Bailey, Pfister, and Henderson, 1978). This is especially important from the resource management standpoint where, e.g., erosion from harvest of a certain forested habitat type may be much greater in a hilly area than in one that is gently sloping. An advantage of regionalization with respect to programs of diversity maintenance not often explicitly mentioned is that occurrences of a major ecosystem type, such as a sandsage-bluestem prairie, in two distinct ecological regions may upon closer examination be found to represent two different ecosystem types.

## Taxonomic Components Approach

Several examples of classification and mapping of plant communities can be used to illustrate the taxonomic components approach. The SAF (Society of

American Foresters) Cover Type Classification (Eyre, 1980) aggregates similar stands of forest trees on the basis of floristics (taxa, e.g., species) and ecological dominance in terms of basal area. The basic taxonomic unit is a forest cover type, a descriptive classification of forest land based on present occupancy of an area by tree species, either in a pure stand or as a grouping of two or more species. It may be a climax type or a seral stage and it is named after its predominant tree species as determined by basal area or stem counts. (A climax plant community is one that is apparently self-regenerating and relatively stable, whereas seral refers to a younger stage that results from a perturbation and which often advances in a characteristic way over time to a certain climax stage. This progression is called "succession.") Ninety forest cover types are recognized in the eastern United States and Canada and 55 in the west. This classification, which is based on existing vegetation, is commonly used in forest surveys. The Renewable Resources Evaluation Group (RRE) of the U.S. Forest Service has aggregated many of the SAF forest cover types into 20 "major forest types" and presented them in a color-coded map at a scale of 1:7,500,000. This is the only map of forest cover types available for the United States as a whole. The Forest Cover Types Manual (Eyre, 1980) describes each type in considerable detail, including its successional status and tendencies.

The Component Classification System developed recently under a Federal interagency agreement involving the U.S. Forest Service, Soil Conservation Service, Bureau of Land Management, Fish and Wildlife Service, and Geological Survey includes four components: vegetation, soils, landform, and water (Merkel et al., 1980). The vegetation classification is based on climax or near-climax plant communities and is arranged hierarchically (see Appendix A).

The upper four levels from formation class through formation are adapted from the International Vegetation Classification and Mapping System of UNESCO (United Nations Educational, Scientific and Cultural Organization, 1973), which is based on climax or near-climax vegetation and uses physiognomic (outward appearance, e.g., forest or shrub) traits. The fifth or subformation level is based on floristics as well as physiognomy and is named wherever possible by a single dominant genus. A later description of the system (Driscoll et al., 1983) omits the subformation level. The series or sixth level is usually characterized by an individual dominant species, although codominants are permissable. The association or seventh level is represented by plant communities of definite composition with a uniform physiognomy and growing in a uniform habitat (Daubenmire, 1968), and is usually named for a conspicuous dominant species in each vegetative layer from top to bottom.

Although the Interagency Component Classification System is not in common use at present, the hierarchical nature of its vegetation component has been described in some detail because of its similarity to several other systems used by the Forest Service and other agencies, e.g., ECOCLASS (Corliss and Pfister, 1973), modified ECOCLASS (Buttery, 1978), Digitized Systematic Classification for Ecosystems (Brown, Lowe and Pase, 1980), and Vegetation Classification System for California (Paysen, Derby and Conrad, 1982). The lowest levels of the system, series and association, are integral parts of the habitat type classification developed by Daubenmire (1952) which is now used extensively by the Forest Service in the western United States (e.g., see Moir and Hendzel, 1983; Pfister, 1984).

Habitat typing is a strongly taxonomic and inductive methodology (Pfister, 1984). A large number of individual, homogeneous climax or near-climax stands (and sites) are sampled for vegetation and sometimes other variables such as soils. Relatively similar stands are grouped subjectively and by computerized multivariate statistical techniques. Individual species are often ranked or "ordinated" along directional axes or environmental gradients to help define the boundaries of plant associations and evaluate diagnostic species. Each plant association defined in this fashion is then named by a single dominant climax tree species (when applicable) followed by a characteristic undergrowth species, e.g., Pseudotsuga menziesii/Vaccinium caespitosum (Douglas-fir/dwarf huckleberry). All plant associations with the same dominant species in the uppermost layer of the canopy are grouped to form a series, e.g., the Pseudotsuga menziesii Series. Plant associations may also be subdivided into phases that represent minor differences in climax vegetation and environmental conditions. A taxonomic key is then provided to aid users in identifying new sites and the whole system is field-tested and fine-tuned. Habitat type is the name given to an area of land that is potentially capable of producing a particular plant association at climax, regardless of the existing successional stage of vegetation on the site. Mapping of habitat types at the series or association levels can be done by anyone familiar with the taxonomy.

Thirty habitat type classifications pertaining to parts of 12 western states and Canadian provinces have been produced between 1963 and 1981 (Pfister, 1984). Most of these pertain to forest ecosystems. Habitat types are now being developed for non-forest ecosystems (e.g., see Mueggler and Stewart, 1980) and somewhat similar procedures are being used to produce "community type" classifications for vegetation types of uncertain successional status (e.g., see Mueggler and Campbell, 1982; Youngblood, Padgett, and Winward, 1985).

Habitat typing provides a strong foundation for preservation of the biological aspects of natural ecosystem diversity in the western United States at its most basic practical level, that of the plant association (and phase). Lists and maps of habitat types provide detailed inventories of the types of ecosystems for which viable and representative samples need to be maintained. Although numerous taxonomic keys have been developed for habitat types, the associated maps do not yet appear to have been produced in most cases (for an exception, see Deitschmann, 1973).

Küchler's classification and mapping of potential natural vegetation types (Küchler 1964, 1966; U.S. Department of Agriculture, 1978) provides another important example of the taxonomic, components approach. Potential natural vegetation (PNV) is defined by Küchler (after Tüxen, 1956, 1957) as the vegetation that would exist today if man were removed from the scene and if the plant succession after his removal were telescoped into a single moment. This concept is similar to that of climax vegetation in that an area can be described by its PNV regardless of the existing vegetation (or lack of vegetation) on the site (for a discussion of differences in the two concepts, see Schlatterer, 1983).

Küchler identified 116 PNV types which are located on a color-coded map of the conterminous (48) United States with a scale of 1:3,168,000 and described in an accompanying manual. These were reduced to 106 on his revised 1966 map which uses a scale of 1:7,500,000. His system is based primarily on physiognomic traits and secondarily on floristic ones. The naming of types reflects this, e.g., Spruce-cedar-hemlock forest (Picea-Thuja-Tsuga), Southern mixed forest (*Quercus*-*Carya*-*Pinus*), and Nebraska sandhills prairie (*Andropogon*-*Calomovilfa*). While the types are named at the generic level, the manual provides a list of dominant and other component plant species for each type, as well as additional information on physiognomy and geographical distribution.

Küchler's PNV types and maps have been widely used for regional and national planning. They provide the only currently organized description of major above-ground, terrestrial ecosystem diversity that describes the entire United States in reasonable detail with respect to all types of vegetation. They do lack some of the desirable attributes of a more typical taxonomic classification. The PNV types are based on only a few characteristics which are not defined precisely, no taxonomic key is provided, and it is difficult to identify types on the ground (Pfister, 1977; Bailey, Pfister and Henderson, 1978). Inspection of the generic names of the types indicates that, with respect to the hierarchical classifications developed by the U.S. Forest Service, they may include subformations, series, parts of series, or groups of habitat types (Corliss and Pfister, 1973). This had led to a number of attempts to cross-reference the Küchler types to other classification systems (e.g., see Garrison et al., 1977; Burns, 1984; DeGraaf and Bailey, in preparation).

A somewhat different taxonomic components approach to ecosystem classification is now being actively used in various state natural heritage inventory programs. Most of these programs developed from cooperative agreements between individual states and The Nature Conservancy, a national, non-profit organization devoted to the protection of ecologically significant lands and resources (The Nature Conservancy, 1975; The Nature Conservancy

News, 1978, 1985). These programs are very important from the biological diversity standpoint because they cover the full range of diversity (plant/animal species and terrestrial/aquatic communities), systematically classify and inventory states, and benefit from the long-term, institutionalized expertise and support of The Nature Conservancy. Typical examples may be found in the natural community classification systems of Pennsylvania (The Nature Conservancy, 1983), Vermont (The Nature Conservancy, 1984a), and West Virginia (The Nature Conservancy, 1984b). Two hierarchical levels (see Appendix B) are typically emphasized, the natural community type and natural community. The latter is defined as a distinct and reoccurring group of organisms naturally associated with each other and their physical environment. It is characterized and defined by a combination of physiognomy, species composition, topography, soil moisture/reaction, and substrate, and named for its most characteristic biotic or abiotic feature. It is the smallest unit of land or water than can be mapped using state natural heritage inventory techniques. (After White, 1978; Haefner, 1982). There may be more than one climax or seral plant community in, e.g., a terrestrial natural community. Lists of dominant and characteristic plant and/or animal species associated with each natural community may be provided for further refinement (e.g., see The Nature Conservancy, 1983).

There are 38 current and one temporarily inactive Nature Conservancy state heritage programs plus those in the Tennessee Valley Region (TVA), the Navaho Reservation, and Puerto Rico. There are also two regional task forces, in the Rocky Mountains and in the Eastern United States, that are attempting to use information from their respective state heritage programs to synthesize regional classification systems and data bases.

Cowardin et al. (1979) developed a taxonomic components type of aquatic ecosystem classification that has quickly attained general acceptance. Wider use of this methodology for inventories of aquatic ecosystem diversity can be expected to produce a considerable increase in the number of representative aquatic ecosystem types that need to be preserved in the United States.

The units of classification used by Cowardin et al. are arranged hierarchically as shown in Appendix C. The highest level, system, is represented by five types: marine, estuarine, riverine, lacustrine, and palustrine. A total of eight subsystems are included in the various systems and 11 classes in the various subsystems. Class describes the general appearance of the habitat with respect to the dominant life form of vegetation (e.g., Emergent Wetland) or physiognomy and composition of the substrate (e.g., Rock Bottom). The dominance type, which is the lowest level in the hierarchy, is determined by the dominant plant species (e.g., Bald Cypress) or the dominant sedentary or sessile animal species (e.g., Freshwater Sponge).

The number of different terrestrial and aquatic ecosystem types ("components") recognized in the classification taxonomies surveyed for this report provides a very preliminary estimate of the magnitude of the natural ecosystem preservation problem in the United States. Information obtained from 16 state heritage programs gave an average of 64 vegetated and non-vegetated terrestrial natural communities per state (forests, grasslands, vegetated wetlands, cliffs, caves, etc.). Information obtained from 15 state heritage programs gave an average of 19 aquatic natural communities per state (lakes, streams, subtidal, marine, etc.). The relatively small number of aquatic natural communities per state quite likely reflects the previous lack of attention biologists have given to the development of classification techniques for aquatic communities. Since several plant associations may be present in each vegetated community (p. 10), it is interesting to consider the eight states which either provided direct information on currently recognized plant associations or information from which such estimates could be deduced. This sample produced an estimate of 223 plant associations per state. Most states included both climax and seral types in their lists. This is appropriate because the majority of these states are east of the 100th meridian, where a general lack of large, representative stands of climax vegetation requires that preservation programs include seral as well as climax plant communities.

West of the 100th meridian copies were also obtained from the U.S. Forest Service and cooperating organizations of 14 different habitat type lists that describe diversity in terms of climax or near-climax plant series and/or associations. These data were less informative because they usually pertained only to parts of states and were mostly restricted to forested ecosystems. The available information suggested very preliminarily that California may have more than 200 plant series and Arizona and New Mexico combined more than 50. With respect to forest vegetation alone, Utah and Idaho may each have more than 20 plant series and more than 100 to 150 plant associations, respectively.

Available information from both the eastern and western United States suggests, then, that an average state may contain more than 200 vegetated terrestrial ecosystems (at the plant association level) that need to be protected. In addition, more than 100 (perhaps several hundred) aquatic ecosystems (at the dominance type level of Cowardin et al., 1979) and nonvegetated above- and below-ground terrestrial ecosystems must be added. All of these types should be protected in a comprehensive preserve system.

## Regionalization Approach

Classification of resources by regionalization was mentioned earlier as an alternative to taxonomic classification of environmental components such as species, communities, soils, and landforms. Actually the two are complementary and can be integrated. Before discussing integration techniques, the methodology of regionalization and some examples will be considered. Regionalization differs from taxonomy in that every classification unit is a unique piece of the landscape with a specific geographic designation. Dasmann (1973, 1974) suggested an international conservation system consisting of a set of world biogeographical regions subdivided into sets of biotic provinces. Udvardy (1975, 1984) continued this effort to devise a satisfactory classification of the world's biotic areas for conservation of species and ecosystems. The highest level in his system is that of a biogeographical realm, a continent- or subcontinent-sized area with unifying features of geography, fauna, and flora (Palearctic, Neotropical, etc.). Subdivisions of realms are called biogeographical provinces. For example, the Nearctic Realm contains two large biogeographical provinces in the eastern United States, the Eastern Forest and the Austroriparian. Udvardy's biogeographical provinces are especially important from a U.S. perspective because they are being used as the main geographical basis for selecting U.S. biosphere reserves (see Fernald et al., 1983 and pp. 23-24 of this report).

Hayden, Ray and Dolan (in preparation) are developing a biophysical coastal and marine classification system for the world, which will be used as a basis for selecting U.S. coastal biosphere reserves (Ray et al., 1981), and preliminary maps of 17 biogeographical coastal provinces in North and South America have been provided (Ray, Hayden and Dolan, 1984).

Commonly used North American regionalizations based on climate and on physiography (physical geography, including land surface forms) are those of Thornthwaite (1931) and Fenneman (1928), respectively.

Examples of U.S. regionalizations that emphasize terrestrial features are Land Systems (Wertz and Arnold, 1972), Ecoregions (Bailey, 1976, 1978, 1980), and Land Resource Regions and Major Land Resource Areas (U.S. Department of Agriculture, Soil Conservation Service, 1981). Regionalizations of forested areas include the widely used Deciduous Forest Regions of Braun (1964) for the eastern United States and Forest Habitat Regions developed by Hodgkins (1965) for the southeastern United States.

Bailey's Ecoregions classification was developed by expanding the work of Wertz and Arnold from earlier concepts of Crowley (1967). It has been used extensively by the U.S. Fish and Wildlife Service, Forest Service, and other Federal land managing agencies. The classification hierarchy (Bailey, 1982, 1983) is shown in Appendix D.

Regional ecosystems (ecoregions) are included in the upper levels of the Bailey hierarchy from domain through district. The levels below district involve more detailed and homogeneous descriptions of landforms, soils, and vegetation. Landtype is at the level of the habitat type of Daubenmire (1952). Ecoregion sections are important with respect to biological diversity because they are identified by single climax vegetation types involving generalizations of Küchler's PNV types (for cross-references between the Bailey and Küchler types, see DeGraaf and Bailey, in preparation).

Bailey's 1976 map of ecoregions of the United States is color-coded, has a scale of 1:7,500,000, and depicts ecoregions down to the level of sections. A descriptive manual accompanies the maps (Bailey, 1978, 1980). In addition, maps of ecoregions and land surface forms of the conterminous 48 states, and of Alaska and Hawaii, are available in color at a scale of 1:5,000,000, with ecoregions down to the level of districts (Bailey and Cushwa, 1982).

The Corvallis Environmental Research Laboratory of the Environmental Protection Agency (EPA) has produced a color-coded ecoregion map of the United States at a scale of 1:7,500,000 that divides the conterminous 48 states into 76 ecoregions based on joint consideration of four primary factors: soils, Küchler's PNV types, Hammond's land surface forms, and land use (Omernik, 1985). All four criteria are used to determine the ecoregion boundaries, although some are given more weight than others in certain areas depending on the varying accuracy of, and information provided by, the individual factor (component) maps. This procedure differs from that of Bailey in that the classification is not hierarchical (i.e., it has only one level), no one factor is emphasized throughout the mapping area (Bailey uniformly emphasized different factors at different levels), and existing land use is one of the factors.

### Integrated Approach

Various types of integrated ecosystem classification schemes can be built from a combination of taxonomic components and land regionalizations. Two different types of examples were discussed by Bailey, Pfister and Henderson

(1978). The first involved the use of ECOCLASS (see p. 6) to integrate desired hierarchical levels from a taxonomic and a regionalization classification in order to identify relatively homogeneous units of land or water. The second occurs commonly in multifactor regionalization schemes such as Bailey's ecoregion provinces (see pp. 14-15 and Appendix D), in which different types of factors (components) may be emphasized at different levels of the hierarchy. For example, Bailey uses Küchler's potential natural vegetation to regionalize at the section level and Hammond's land-surface forms to regionalize at the next lower level of district.

Bailey, Pfister, and Henderson do not discuss a slightly different type of integration which appears to have considerable utility with regard to preservation of biological and ecosystem diversity. This involves a combination of regionalization at one level in the classification scheme with a taxonomic components approach at a lower level in the same classification hierarchy. Three examples will be cited. The first involves the combination of Bailey's ecoregion provinces with Küchler's PNV types used by the U.S. Forest Service in its RARE II analysis of potential wilderness sites (U.S. Department of Agriculture, Forest Service, 1978; Davis, 1984). This produces 242 major ecosystem types (later reduced to 233), each of which can be defined as a PNV type within an ecoregion province; e.g., an Oak-Hickory Forest PNV type in the Eastern Deciduous Forest Province is a different ecosystem from an Oak-Hickory Forest PNV type in the Southeastern Mixed Forest Province. This recognizes that the two plant communities may well be parts of different ecosystems as a result of their long-term existence in the separate environmental regimes of the two ecoregions. It has the further advantage of describing the major ecosystem diversity of the United States in approximately twice as much detail as the 124 PNV types, yet this level of diversity can still be relatively easily comprehended on a map at a scale of 1:7,500,000.

The Integrated Habitat Inventory and Classification System (IHICS) of BLM (U.S. Department of the Interior, Bureau of Land Management, 1978) regionalizes the land base primarily according to Fenneman's (1928) physiographic regions and by subphysiographic regions when more specificity is desired. Immediately below this level, the classification uses Küchler's PNV types. A unit of Juniper-Pinyon Woodland in the Lower Basin and Range Province is then considered different from one in the Upper Basin and Range Province. As the BLM manual states, this is "based on the assumption that the ecosystem determinants used to separate physiographic regions would likewise cause at least subtle, if not profound, differences in plant and animal compositions at the association level." (Author's note: "association" as used in the BLM manual refers to the Küchler PNV types.)

The IHICS System is also noteworthy because its data bases are being linked with a geographic information system capability which will eventually permit rapid determination of the numbers, sizes, and spatial relationships of ecosystem type occurrences in each of the 13 western states (Cooperrider, 1985). The basic taxonomic component in the system is the habitat site, a local, relatively homogeneous ecosystem named for its dominant existing vegetation and landform, e.g., a Low Sagebrush-Silver Sagebrush-Mesa or a Pinyon-Juniper-Canyon. Habitat sites can also be subdivided based on special habitat features such as access to water or presence of cliffs, which make them especially suitable for certain types of wildlife (U.S. Department of the Interior, Bureau of Land Management, 1978).

Cowardin et al. (1979) recommended the use of Bailey's classification (1976) and map (1978) to satisfy the need for inland regionalization of the wetland components in their classification system (see p. 11 and Appendix C) and proposed their own set of 10 regionalized provinces for use with their marine and estuarine wetland components. With respect to the latter, an example of a specific wetland ecosystem located on the southern tip of Florida might be a Broad-leaved, Evergreen (e.g., Mangrove), Scrub-Shrub Wetland in an Intertidal Estuary in the West Indian Province. (The eastern U.S. coastal provinces of Cowardin et al. are rather similar to those proposed by Ray, Hayden and Dolan, 1984, for the same area).

## III. Land Designation for Maintenance and Protection of Ecosystem and Biological Diversity

Two broad classes of land will tend to have designations that offer some degree of protection to ecosystems and biological diversity: (1) public lands and (2) private lands that are dedicated to some aspect and level of protection. A public land unit may be dedicated in entirety but often it will simply contain one or more subunits that are afforded some level of protection. Essentially all public lands that contain natural features are of interest with respect to diversity conservation, regardless of designation.

Federal land designations are the easiest to describe and will be mentioned first. Three references especially useful in helping to organize this information were: The Nature Conservancy's description of Federal activities involving natural area programs (Wieting, 1977), a recent survey of U.S. activities related to in situ conservation of genetic resources (U.S.

Department of State, Bureau of Oceans and International Environmental and Scientific Affairs, 1984), and a new map with accompanying text on U.S. networks of ecological research areas (U.S. Department of the Interior, Geological Survey, 1985). In addition, recent information not referenced was obtained from various agency publications. Abbreviated definitions and descriptions, as well as approximate statistics, are used in the discussion that follows.

The land units of special interest in the National Park Service are: National Park (covers large enough area to ensure adequate preservation of a variety of natural resources), National Monument (preserves at least one nationally significant resource, e.g. Black Canyon of the Gunnison River), National Preserve (preserves special resources, e.g., Big Cypress and Big Thicket), National Lakeshore and National Seashore (preserve shoreline areas and off-shore islands), National River/Wild River/Scenic River (preserve ribbons of land along rivers and streams), National Recreation Area (preserves rural and urban lands for recreation), National Parkway (preserves ribbons of land along roadways), and National Scenic Trail (preserves ribbons of lands along trails and is non-motorized).

Most National Park Service additions are by specific acts of Congress, but the President has authority under the Antiquities Act of 1906 to proclaim national monuments on lands already under Federal jurisdiction. As with other land managing agencies, the level of protection for natural diversity will vary, even within units, according to the unit management plans which are themselves periodically updated (e.g., every 10 years). The Park Service recognizes four major categories of management and use with regard to natural resources and processes in the "Natural Zone": Outstanding Natural Feature, Natural Environment, Protected Natural Area, and Research Natural Area. All of these require that the natural resources and processes not be adversely affected and the latter two usually involve relatively undisturbed resources and minimal human access. Wilderness Areas, Wilderness Study Areas, and National Natural Landmarks are also included in the Natural Zone Category.

Forest Service Lands are classified as: National Forest (established by Congress and permanently reserved for multiple uses such as timber, grazing, water, wildlife, and recreation), National Grassland (designated by Secretary of Agriculture and held by Department of Agriculture under Bankhead-Jones Farm Tenant Act), Land Utilization Project (designated by Secretary of Agriculture for conservation and utilization under Bankhead-Jones Farm Tenant Act), Purchase Unit (designated by Secretary of Agriculture or previously approved by National Forest Reservation Commission for purposes of Weeks Law Acquisition), and Research and Experimental Areas (dedicated and reserved by Secretary of Agriculture for forest or range research and experimentation).

The Forest Service also recognizes the following Special Designated Areas: National Forest Wilderness (designated by Congress as part of National Wilderness Preservation System), National Forest Primitive Area (designated by Chief of Forest Service and administered like Wilderness Area pending study of its suitability for Wilderness designation), National Wild River/Scenic River (designated by Congress as part of National Wild and Scenic River System), National Recreation Area (established by Congress to assure protection of, and provide management for, public outdoor recreation), Cascade Head Scenic - Research Area (established by Congress for use, enjoyment, study, and protection of certain ocean headlands), National Game Refuge and Wildlife Preserve (designated by proclamation or by Congress for protection of wildlife), and National Monument (object of historic or scientific interest, declared by proclamation or by Congress). Forest Service lands are also included in the National Natural Landmarks and National Trails programs.

The goals of the U.S. Fish and Wildlife Service directly related to diversity conservation are to preserve, restore, and enhance all species of threatened or endangered animals and plants in their natural ecosystems (when practicable), to perpetuate the migratory bird resource, and to preserve the natural diversity and abundance of animals and plants in the National Wildlife Refuge System. The Service also identifies significant wildlife resource areas on a state-by-state basis (e.g., see U.S. Department of the Interior, Fish and Wildilfe Service, 1981) and designates "critical habitats" under authority of the Endangered Species Act of 1973. Certain lands within the National Wildife Refuge System are designated as Wilderness Areas, Research Natural Areas, and National Natural Landmarks. Since the main purpose of a refuge may be to preserve and enhance a particular wildife species or species group such as elk or waterfowl, certain types of relatively intensive management may be practiced that especially benefit those species. Examples of these specialized management techniques are ones that are applied to marshes and water, grasslands, forests, and croplands. Some refuges are, however, maintained in a relatively natural, undisturbed state for various reasons. As with all of the land managing agencies, detailed information on protection levels associated with individual units must be obtained from the management plans of the units.

The Bureau of Land Management (BLM) administers in excess of 300,000,000 acres of mostly multiple-use public lands through State, District, (within states), and Resource Area (within districts) Offices. Its Natural History

Resource Program provides for selection, establishment, and management of Research Natural Areas, Outstanding Natural Areas, and Areas of Critical Environmental Concern. These areas are all designated internally by the agency, following established procedures. BLM also designates lands for special recreational use (Recreation Lands), and BLM lands contain Wilderness Areas, Primitive Areas, National Trails (Historic, Scenic, Recreational), and National Natural Landmarks.

The Department of Defense is another large Federal land managing agency. The Department's directives require all military installations and other holdings to be managed for multiple uses while protecting, conserving, and managing the plants, animals, and natural landscapes as integral ecosystem elements. Besides military installations, the Department manages significant amounts of land through its Army Corps of Engineers in the form of water resource development projects. The Corps has a number of internal land designations that pertain to different densities of recreational use, natural areas, and forest and wildlife management.

Other Federal agencies that manage significant amounts of land and water resources or smaller holdings of special ecological value include the Bureau of Indian Affairs, Bureau of Reclamation, Department of Commerce, and Department of Energy. Bureau of Reclamation lands and waters are acquired and managed in conjunction with its water development and reclamation projects in the western United States. The areas are either withdrawn from the public domain, acquired, or held in the form of easements. They are managed for multiple uses and recreation is often emphasized.

The Department of Commerce is authorized under the Coastal Zone Management Act of 1972 to make grants to coastal states for development of

Estuarine Santuaries and under the Marine Protection, Research, and Sanctuaries Act of 1972 to designate Marine Sanctuaries. The latter may or may not involve waters under the jurisdiction of a state, and include seaward, coastal tidal zones, and Great Lakes waters. Estuarine Sanctuaries are not required to have legal protection and may be altered or disestablished by states. Marine Sanctuaries provide recognition and additional protection according to the specifics of Federal-state agreements. Off-shore drilling is not prohibited unless specifically stated in the agreement. The National Marine Fisheries Service of the Department administers the program. In 1984 there were six Marine Sanctuaries, with several other sites under consideration.

The Department of Energy has established five National Environmental Research Parks for studies on, or involving, the monitoring of environmental effects of human activities and predicting and mitigating environmental impacts. Public access to the natural ecosystems under study is generally limited. An important part of the overall program at each site involves establishing reference areas to serve as reservoirs of genetic diversty as well as experimental controls.

There are five national networks of ecological research areas in the United States, some of which have been mentioned previously. Biosphere Reserves are part of an international network designed to conserve the diversity and integrity of representative natural ecosystems in the world's major biogeographical provinces, furnish large areas for environmental baseline studies and research, and offer educational study opportunities. Protection is provided through voluntary agreements with managers of existing land units, either public or private. Most Biosphere Reserves are large and many exceed 100,000 acres. As of January, 1984 there were 40 Biosphere Reserves in the United States, ranging in size from 150 to 7,600,000 acres. The five National Environmental Research Parks of the Department of Energy, which were mentioned in the preceding paragraph, vary in size from 12,500 to 572,500 acres. There are 88 Experimental Forests and Ranges designated and operated by the U.S. Forest Service to provide sites for manipulative research. They vary in size from 170 to 28,000 acres. Experimental Ecological Reserves are designated by the National Science Foundation and used for long-term experimental field research. There are 96 sites which range in size from 175 to 27,500 acres. Research Natural Areas form the largest and oldest group of ecological research areas. Their purposes are to protect the full range of natural communities and biological diversity in the United States and to provide sites for educational use and nondestructive research. There are 375 sites distributed among lands of the Forest Service, BLM, Fish and Wildlife Service, Park Service, Department of Energy, Department of Defense, and TVA. They are designated by these agencies and are usually larger than 300 but less than 2,500 acres. However, the range is from 2 to 840,000 acres! Over 140 are on Forest Service lands. Not all of the various ecological research areas just described are separate units, as some carry more than one designation. For example, the Hubbard Brook property of the Forest Service in New Hampshire is an Experimental Forest, a Biosphere Reserve, and part of an Experimental Ecological Reserve.

National Natural Landmarks are important examples of U.S. natural history that contain nationally significant ecological or geological features. The Landmarks Program was administratively created within the Department of the Interior. The Secretary identifies landmarks which may also be formally registered as such through agreements between the Secretary and the landowner. Official recognition and voluntary registration are the only direct protection provided. Many landmarks are managed by public agencies and many are privately owned and managed.

Inspection of information obtained in 1985 from a large number of state heritage and natural area programs indicates the following sorts of designations applied by state and local governments to their owned and/or managed natural lands: states - park, forest, wildlife area, fishing area, natural area, preserve, conservation area, wilderness, experiment station, university research station or area, botanical site, recreation area, school and other public lands, special district (e.g., water management district within part of a state); counties and municipalities - park, open space, preserve, forest preserve, conservation district, wildife santuary, greenbelt. Many of these state and local lands are publicly owned. Management varies, even within one type of category and specific local government. For example, the City of Boulder, Colorado, which owns (or in a few cases, leases) 20,000 acres of mountain parks, reservoirs, and open space, allows livestock grazing in several areas and restricts human access in a few, limited areas (Crumpacker, 1985). Local governments may also zone private lands in ways that provide different levels of protection for natural resources such as wildlife habitat.

A private organization, the Society of American Foresters (SAF), designates and registers forested natural areas (Buckman and Quintus, 1972). Their system overlaps the Research Natural Area network established on Federal lands but also includes state, local, and privately owned sites. The SAF natural areas are maintained in as near a natural condition as possible, usually without direct human intervention. They are intended to provide reservoirs of genetic diversity for the different SAF forest cover types of the United States and Canada, as well as sites for scientific observation and education. SAF natural areas are generally 300 to 1,000 acres in size and there are approximately 500 sites in the system.

Several private organizations own and manage significant amounts of ecologically important natural areas in the United States, e.g., The Nature Conservancy (over 700 privately held preserves), National Audubon Society, State Audubon Chapters, and The Trustees of Reservations in Massachusetts. In addition, there are many small trusts that manage natural areas for ecological, scenic, or open space purposes. Some of these privately owned preserves and sanctuaries are among the most ecologically important natural areas in the country and nearly all are among the most highly protected.

Assigning levels of protection for ecosystem and biological diversity to the various land designations described in this section is a complex problem. Two important considerations are the type of protection indicated by the designation (National Park, BLM Public Lands, Research Natural Area, State Forest, County Open Space, Private Nature Preserve, etc.) and the strength of the designation (Act of Congress, administrative designation by the U.S. Forest Service, voluntary registration by a private party, purchase by a private trust, etc.). The highest level of protection appears to be that of privately owned and legally secured, single-purpose, nature reserves. Publicly owned areas designated by legislative bodies and dedicated wholly or largely to preservation of at least some subunits (wildernesses, national parks, national monuments, some municipal open space, certain wildife refuges, etc.) appear to come next. The least protected areas are those which depend on voluntary cooperation of the owners and managers (e.g., national natural landmarks and many state natural areas), or which do not emphasize diversity conservation and have other ascendant priorities (e.g., Federal military installations and state school lands). Most designated areas lie somewhere between these extremes, including the vast multiple use and variously designated lands of the U.S. Forest Service and Bureau of Land Management. Even the actual level of protection provided to a research natural area by, e.g., an individual national forest is not easy to assess fully and may depend on many factors such as its history of use (Franklin, 1984), the land and resource management plan of the forest, the dedication of forest personnel to the plan, the local economy, and administrative guidance at the regional and national levels. (For additional discussion of general types of management and protection categories, see Fernald et al.,1983; IUCN's Commission on National Parks and Protected Areas, 1984; and Goodland, 1985).

## IV. Current Protection Status of U.S. Ecosystems

The extent to which U.S. ecosystem diversity is represented on lands that offer some degree of protection is being assessed on a state-by-state basis by the natural heritage inventory programs of the different states. As mentioned earlier, few of these surveys consider terrestrial ecosystems at the detailed level of plant association, although this type of information may be available at some future time.

At the level of Küchler's PNV types, which can be considered to be representative of major U.S. terrestrial ecosystem types, Klubnikin (1979) surveyed 280 designated natural areas in 10 statewide park and preserve systems in California. She found that 46 of the state's 53 PNV types included in her analysis (87%) were represented somewhere in this collection of natural areas. Five (9%) were greatly overrepresented, having more than 50% of their areal extent included, and 22 (42%) were greatly underrepresented, with less than 3% of their area included. On a national level, Armentano, Smith and Loucks (1981) surveyed the representation of ecosystems (mostly Küchler PNV types) in the National Science Foundation network of experimental ecological reserves for the United States and Puerto Rico and found that 84 of the 133 types (63%) were included. The California parks and preserves studied by Klubnikin and the experimental ecological reserves just mentioned average about 4,600 and 15,000 acres, respectively, and some are quite small. A number of these sites may, therefore, not be large enough to maintain the full range of diversity typical of many PNV types and their respective successional stages without fairly intensive management, even if the climax types are actually represented in the existing vegetation.

Crumpacker (1979) made a preliminary survey of the protection accorded to Küchler's PNV types nationwide in units of approximately 23,000 acres or larger and obtained more disturbing results. He used maps A and B originally prepared by the U.S. Geological Survey at a scale of 1:7,500,000 for use in the U.S. Forest Service's RARE II analysis of potential wilderness areas (U.S. Department of Agriculture, Forest Service, 1978). Map A shows the principal lands administered or held in trust by the National Park Service (NPS), U.S. Fish and Wildlife Service (FWS), U.S. Forest Service (FS), and Bureau of Land Management (BLM). Map B describes major ecosystems of the United States and Puerto Rico in terms of Küchler PNV types within Bailey ecoregion provinces. The two maps were overlaid to determine which ecosystem types were not adequately "protected" by the four agencies (see Appendix E for detailed presentation of results). Of the 315 ecosystem types identified, 91 (29%) had only minimal Federal protection, i.e., they occurred in only one unit of the NPS, FWS, or FS (e.g., in only one park and nowhere else) or only on BLM lands (considered minimal because of the large amount of private inholdings not shown inside the BLM boundaries on map A); 69 others (22%) had no definite Federal protection, i.e., did not occur in any of the four agency's holdings. The majority of minimally protected ecosystem types (49, or 54%) were on BLM lands (see Appendix F for more detailed analysis). The largest numbers of ecosystem types with no definite Federal protection were in Texas and Oklahoma, which therefore offered the greatest opportunities among the states for filling gaps in the U.S. "macro-reserve" system (see Appendix G for detailed analysis).

Since Crumpacker's analysis assessed only potential diversity and did not attempt to determine the extent to which the Küchler types actually occur on the landscape in any successional stage, it is likely that the true ecosystem coverage is less, perhaps considerably less, than his preliminary data indicate. Furthermore, the level of protection accorded to these ecosystems, even if they are actually represented in the agency lands, is presently unknown (although it could presumably be determined).

Crumpacker is currently attempting to expand his analysis by updating it and extending it (1) to include land holdings of other Federal agencies and of state and local agencies and private organizations; (2) to all parcels of 5,000 acres or larger (instead of his earlier minimum of 23,000 acres); (3) to include the relationship between potential and existing ecosystem diversity; and (4) to include the level of protection actually provided each natural area by its manager. With regard to item 4, a questionnaire is currently being assembled with aid from the U.S. Man and the Biosphere Program that will ask land managers (Federal, state, etc.) to provide anonymous assessments of how well a wide range of management objectives are being met on their respective units. These objectives will include a subset related specifically to protection of species and ecosystem diversity.

Davis (1984) conducted an analysis somewhat similar to that of Crumpacker, using the Forest Service's RARE II approach to definition and classification of major ecosystem types, which he had been instrumental in developing (Davis, 1980, 1981). His criteria for identification of ecosystem types on RARE II map B were more stringent, since he excluded Küchler PNV types that might not in fact have existed in separate ecoregions (and thereby have qualified as separate ecosystems) if Bailey's ecoregion province boundaries could have been drawn more exactly on a map of larger scale than 1:7,500,000 (see Appendix H for detailed results). He found 50 ecosystem types out of 233 (21%) that did not appear to be represented in any holdings of the four agencies (NPS, FWS, FS, BLM). These would appear to be ecosystem types similar to those that Crumpacker (1979) described as having "no definite Federal protection." Hence the independent analyses by Davis and Crumpacker produced virtually identical results with respect to proportion of unprotected major ecosystem types.

Davis's 1984 analysis is also interesting because he used it to assess major ecosystem gaps in the National Wilderness Preservation System (NWPS) of the United States (as he had done previously in the RARE II study; see Appendix I for detailed results). Using a criterion of at least two examples of 1000 acres each of an ecosystem type for adequate representation, he found that 81 of the 233 types (35%) were adequately represented in the NWPS and that 102 could potentially be brought to this status by additions from existing agency holdings. The remaining 50 mentioned earlier, many of them in the grassland areas of the Midwest, would have to come from new Federal land acquisitions or new state wilderness programs.

## V. Assessment of Changes in Ecosystem Diversity

Natural ecosystem diversity is obviously decreasing in the United States (Klopatek et al., 1979). Between 1967 and 1975, forest land in the conterminous 48 states declined by approximately 70,000,000 acres while, as a result of balanced amounts of additions and losses, cropland remained relatively unchanged at approximately 438,000,000 acres (Dideriksen, Hidlebaugh and Schmude, 1977). Although cropland represents the single most important source of potential replacement land for natural terrestrial ecosystems, it is increasingly being lost to urban areas, transportation corridors, and other installations which are expected to grow in size from 100,000,000 acres in 1970 to 122,500,000 acres in 1990 (Diderikson, Hidlebaugh and Schmude, 1977).

Klopatek et al. (1979) have produced interesting estimates concerning how much land in the United States is still capable of being occupied by natural vegetation in various successional stages. Their approach involved the following steps: (1) information on county locations within the conterminous 48 states was used in conjunction with Küchler's 1964 PNV map (1:3,168,000), updated to include the types listed on his 1966 map, to estimate the area in each county originally occupied in presettlement times by each PNV type; (2) the 1967 Conservation Needs Inventory (CNI) land use data (U.S. Department of Agriculture, 1971), which were the most recent land use data available at a county level, were aggregated to give estimates of lands not now occupied by forest or rangeland; (3) based on various considerations, each PNV type was assigned a probability of having been converted to one of these non-natural land cover types in order to estimate how much of each PNV type per county (if there were more than one) had in fact been converted; and (4) the non-naturally vegetated county land use totals were subtracted from their respective PNV types to estimate the amount of PNV types(s) currently remaining in each county. All of the PNV and CNI data used in this analysis are part of the county-level environmental data base for the counterminous United States maintained at the Oak Ridge National Laboratory in Oak Ridge, Tennessee (for the latest compilation, see Olson, Emerson and Nungesser, 1980).

Klopatek et al. then ranked each PNV (ecosystem) type by its estimated % change from presettlement times. Values ranged from -89% to -1%. The Tule Marsh PNV type in California, Nevada, and Utah showed the greatest loss, 89%, primarily as a result of agricultural encroachment. Next in line was the Elm-Ash Forest type of Ohio, Indiana, and Michigan with an 88% decrease, due principally to agriculture and urbanization. Vegetation in the western United States showed the least losses, with a few exceptions due to agricultural and urban spread. Twenty-three PNV types that once covered approximately 50% of the conterminous 48 states now cover 7% at best. The agricultural states of Iowa, Illinois, and Indiana have lost the highest amounts of their natural terrestrial ecosystems (92, 89, and 82%, respectively). States with the least loss were Nevada, Arizona, and New Hampshire (4, 7, and 12%, respectively). These estimates were based on changes that have occurred over 100 or more years (usually more, depending on location) from presettlement times to 1967.

(Author's note: An update of the analysis using data from the late 1970's would provide an estimate of change over the most recent decade for which data are, or will soon be, available).

Klopatek et al. had to use potential instead of existing vegetation information on which to base ecosystem types because it was all that was available. This left out some nationally important ecosystem types such as aspen, birch, and certain riparian and wetland areas which are not included in the Küchler PNV types. (A large amount of remotely sensed existing vegetation data have been produced by the military but are classified and unavailable; Hamlin and Goodenough, 1978). It is important to note in this regard that Hoekstra and colleagues at the Rocky Mountain Forest and Range Experiment Station in Fort Collins, Colorado are developing an automated information system for aggregation of county-level data throughout the United States that is based on detailed ground inventories of existing vegetation, soils, and other parameters (Chalk, Miller and Hoekstra, 1984). These survey data provide a basis for the multiresource inventories required of the U.S. Forest Service and Soil Conservation Service by, respectively, the Forest and Rangeland Renewable Resources Planning Act of 1974 (RPA; for forest and rangeland resources) and the Soil and Water Conservation Act of 1977 (RCA; for soil, water, and related resources on nonfederal lands). A key feature of the information system approach (Chalk et al.) involves a series of hierarchical matrices that can be used to relate land use categories such as rangeland and forest land to land cover categories such as grassland, shrubland, and treeland. The land cover categories can then be related to ecological categories such as the Forest-Range Ecosystem Types used in RPA summaries (mountain grasslands, desert shrub, oak-pine, etc.) which, in turn, can be cross-referenced to Küchler's PNV types (Garrison et al., 1977).

Since the RPA and RCA assessments are required by law to be conducted every 10 years, they will provide a nationwide basis for estimating rates of change in the amount of existing terrestrial (and aquatic) ecosystem types. Furthermore, they can be linked to econometric models that project land use changes in order to predict future changes in natural ecosystem types (e.g., see Hoekstra et al., 1983).

The U.S. Fish and Wildlife Service is actively inventorying U.S. wetlands. Current wetland status and both recent and historical trends in major wetland ecosystem types have been discussed by Tiner (1984). Total wetland acreage in the conterminous United States is estimated to have been 215,000,000 in presettlement times, 108,000,000 in the mid-1950's, and 99,000,000 in the mid-1970's. The total loss from presettlement times to the mid-1970's was 54% and over the 20-year period between the mid-1950's and mid-1970's it was 8%. Drainage for agricultural purposes has been the main factor. Tiner also described eight current national wetland problem areas.

From 70 to 90% of the natural riparian (largely streamside) vegetation in the United States is believed to have been lost to human activities in the 19th and 20th centuries (U.S. Council on Environmental Quality, 1978; Warner, 1979a; Swift and Barclay, 1980). Losses of 98.5% have been estimated for the Sacramento Valley of California (Smith, 1980), and of 95% or more for Arizona. Large losses in wildlife density and diversity must have accompanied these changes, since western U.S. riparian ecosystems provide luxurious habitats in comparison with adjacent uplands (see Crumpacker, 1984 for additional discussion and references). Most of the remaining riparian areas in the United States are thought to be in an unsatisfactory condition due to human-related impacts (Almand and Krohn, 1978; Warner, 1979b). Most riparian ecosystems are too small to be included as potential natural vegetation on a map of scale smaller than 1:3,000,000 and therefore cannot be included in analyses such as that of Klopatek et al. (pp. 31-33).

An alternative means of assessing rates of natural ecosystem conversion and change involves the use of remote sensing techniques. This requires the archiving of valuable remotely sensed data collected in the past, as well as the use of improved methods for surveying vegetation that are now being developed.

Improved satellite methods of surveying major vegetation types of whole continents at the physiognomic level of formation are now available (Tucker, Townshend and Goff, 1985), but satellite information on floristics of large areas (e.g., at the series level) is still rather inaccurate (see Lindauer, Williamson, and DePlazes, 1984). Medium scale color infrared photography offers better floristic resolution (Lindauer et al.) and is being used by BLM to slowly map the existing vegetation of its priority lands in the western United States. Major studies are being initiated by the National Aeronautics and Space Administration and other organizations to analyze land use changes from the last 10 years of satellite data, and new radar and laser techniques offer promise for more detailed analyses of certain types of vegetation such as those typical of wetlands (Wickland, 1985).

## VI. Summary and Information Needs

Many natural ecosystem classifications are currently being used in the United States. The state heritage, U.S. Forest Service, U.S. Soil

Conservation Service, and Bureau of Land Management programs are especially important because they are building systems and data bases that will eventually provide regional and national perspectives. More effort needs to be put into classification at the basic level of plant associations in terrestrial ecosystems and the dominance types of Cowardin et al. in aquatic ecosystems. Information surveyed for this report suggests that the number of vegetated ecosystems which need to be preserved in representative samples at the plant association level will average more than 200 per state, and this does not include perhaps 100 or more aquatic and nonvegetated natural ecosystems per state. Consideration of ecoregion boundaries may lead to identification of additional ecosystems. A major effort should be made to organize the widely scattered information base for successional stages that lead to different plant association climaxes, since successional types must also be included in the national preservation system, either as parts of climax ecosytems or as separate units (the latter will be necessary in many parts of the eastern United States and possibly in various western areas where the native shrublands and grasslands have been seriously disturbed by overgrazing).

With respect to vegetated ecosystems, one or more reasonably large and representative samples of each seral, as well as climax, plant association type will need to be protected in many parts of the eastern United States. In the west, large stands of many climax plant associations may still exist. Representative stands of these types, large enough to ensure the presence of all the seral communities associated with each type, will need to be protected. Alternatives to this would include active management of smaller climax stands to provide for presence of desired seral stages or protection and maintenance of separate seral stands. The number of additional nature reserves needed in a comprehensive preservation system for the United States cannot be estimated until nationwide inventories of ecosystem types and their current protection status are closer to completion.

Küchler's potential natural vegetation map of the United States combined with one or more appropriate regionalized maps will provide a reasonable survey of major above-ground, zonal, climax, terrestrial ecosystem diversity for present national planning purposes and should be continually cross-referenced with developing classification systems, data bases, and vegetation maps at more basic levels to improve its accuracy. As mentioned earlier (p. 9), some Küchler types may include up to several plant series or even subformations, and hence several to many climax plant associations. National aquatic, azonal (e.g., vegetated wetlands and riparian), below-ground, and seral ecosystem diversity will have to be surveyed and assessed by other means.

No reasonably comprehensive method exists for describing the different levels of protection currently associated with natural ecosystems throughout the United States. This problem will not be easy to solve for multiple use management areas that have many different management plans and designated subunits, and which are usually exposed to varying levels of pervasive factors such as consumptive and nonconsumptive recreation. A serious attempt should be made to devise a classification scheme that accurately describes the actual protection levels accorded to ecosystem diversity in the Federal, state, and local public domains, and the private domain. The questionaire described on pp. 29-30 is being designed to provide information of this sort.

Documentation of the extent to which potential and existing ecosystem diversity are protected at the Federal, state, local, and private levels should continue to be supported, as should studies of the rates at which natural ecosystems are being lost. These efforts help greatly to prioritize preservation efforts and make them cost effective.

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# APPENDIX A

Vegetation Classification System Used in the Component Land Classification System Developed under the "Interagency Agreement Related to Classifications and Inventories of Natural Resources" (Merkel et al., 1980; Driscoll et al., 1983).

## 1980 Version

| Hierarchical Level | Example |
|---|---|
| Formation Class | Woodland |
| Formation Subclass | Evergreen Woodland |
| Formation Group | Temperate Evergreen Woodland |
| Formation | Temperate Evergreen Woodland with Rounded Crowns |
| Subformation | Western *Pinus* Woodland without Evergreen Schlerophyllous Understory |
| Series | Western Montana *Pinus ponderosa* Woodland |
| Association | *Pinus ponderosa*/*Agropyron spicatum* |

## 1983 Version

- Class
  - Subclass
    - Group
      - Formation
        - Series
          - Association

# APPENDIX B

Typical Example of Natural Community Classification System Used in State Natural Heritage Inventory Programs (after White, 1978; Haefner, 1982).

## Hierarchical Levels

Natural Community Type

Natural Community

| Natural Community Type | Short Definition |
|---|---|
| Marine | Coastal zone of ocean |
| Estuarine | Wetlands under tidal influence |
| Lacustrine | Lakes and ponds |
| Riverine | Rivers and streams |
| Palustrine | Non-tidal wetlands with emergent vegetation |
| Terrestrial | Non-aquatic above-ground areas |
| Subterranean | Below-ground areas |
| Cultural[1] | Created by human disturbance |

## Examples of Natural Communities[2]

Xeric Central Hardwood - Conifer Forest
Dry Southern Oak/Pine Forest on
  Sandy/Gravelly Soils
Central Appalachian Heath Barren
Talus Cave
Calcareous Cliff Community
Moderately Alkaline Lake/Pond
Low-Gradient Clearwater River
Graminoid Marsh

---

[1] May include some completely non-natural types such as Cultivated Cropland and Developed Land.

[2] More than one climax or seral plant community may be included in a natural community. Lists of dominant and characteristic plant and/or animal species associated with each natural community may be provided for additional refinement (e.g., see The Nature Conservancy, 1983).

## APPENDIX C

Wetland and Deepwater Habitat Classification System of Cowardin et al. (1979).

| Hierarchical Level | Example |
|---|---|
| System | Lacustrine |
| Subsystem | Limnetic |
| Class | Rock Bottom |
| Subclass | Bedrock |
| Dominance Type | Freshwater Sponge (*Spongilla lacustris*) |

# APPENDIX D

## Ecoregions Classification of Bailey (1976, 1978).

| Hierarchical Level | Example |
|---|---|
| Domain | Humid Temperate |
| Division | Prairie |
| Province | Prairie Brushland |
| Section | Juniper-Oak-Mesquite |
| District | High Hills |
| Landtype Association[1] | - |
| Landtype[2] | - |
| Landtype Phase[3] | - |
| Site[4] | - |

[1] Part of a district determined by isolating areas whose form expresses a climatic-geomorphic process.

[2] Part of a landtype association having a fairly uniform combination of soils and vegetative succession, e.g., a particular soil series and "Daubenmire" habitat type.

[3] Part of a landtype based on variations of soil and landform properties, e.g., soil drainage and slope, that affect productivity of the landtype association habitat type.

[4] Part of a landtype phase that is homogeneous in respect to all components, including soils and vegetation, and their response to management.

# APPENDIX E

Major Ecosystem Types of the United States and Puerto Rico that have Minimal or No Definite Federal Protection (photocopied from Crumpacker, 1979).[1]

Numbers in parentheses refer to RARE II map B identification codes. Each PNV type within an ecoregion province is a separate ecosystem. One asterisk means the ecosystem has no definite Federal protection; two asterisks means that its PNV type also has no Federal protection in any other ecosystem; three asterisks mean that it also is definitely not a trivial ecosystem. Questions marks mean uncertain; e.g., in numerous instances the relatively small scale of RARE II maps A and B did not permit a definite decision as to whether or not at least part of an ecosystem lay within the boundaries of a particular Federal land unit. The state or states in which an ecosystem is located are identified for all ecosystems that have no definite Federal protection. See text and RARE II map A for abbreviations and other details.

| PNV Type | Federal Managing Agency or Federal Land Unit | State or States in which the Ecosystem is located |
|---|---|---|
| | Arctic Tundra Province (1210) | |
| Alder Thickets (115) | BLM | |
| Dryas Meadows and Barren (118) | BLM | |
| | Brooks Range Province (M1210) | |
| Muskeg (114) | BLM | |
| Alder Thickets (115) | BLM | |
| Watersedge Tundra (117) | BLM | |
| Ice Fields (130) | BLM | |

---

[1] See p. 29 for definition of minimal Federal protection and no definite Federal protection. A non-trivial ecosystem (see para. 1 above) is one that is not associated with a small extension of a larger Küchler PNV type into an adjacent Bailey ecoregion province (see p. 30 for further explanation). Uncertain protection areas (see question marks in tabular information) were not subsequently counted as providing protection .

## Yukon Forest Province (1320)

| | | |
|---|---|---|
| Spruce-Birch Forest (112) | BLM | |
| Black Spruce Forest (113) | BLM, Mt. McKinley NP? | |
| Muskeg (114) | BLM | |
| Alder Thickets (115) | BLM | |
| Cottonsedge Tundra (116) | BLM | |
| Dryas Meadows and Barren (118) | BLM | |

## Alaska Range Province (M1310)

| | | |
|---|---|---|
| Alder Thickets (115) | BLM, Katmai NM? | |
| Watersedge Tundra (117) | BLM, Kenai National Moose Range?, Chugach NF? | |

## Bering Tundra Province (1220)

| | | |
|---|---|---|
| Spruce-Birch Forest (112) | BLM, Clarence Rhode National Wildlife Range? | |
| Black Spruce Forest (113) | BLM | |
| Alder Thickets (115) | BLM | |

## Hawaiian Islands Province (M4220)

| | | |
|---|---|---|
| Schlerophyllous Forest, Shrubland, and Grassland (121) | Hawaiian Volcanoes NP, Haleakala NP? | |
| ***Guava Mixed Forest (122) | Hawaiian Volcanoes NP?, Haleakala NP? | Hawaii |
| ***Lama-Manele Forest (124) | None | Hawaii |
| ***Koa Forest (125) | None | Hawaii |
| Koa-Mamani Parkland (126) | Hawaiian Volcanoes NP, Haleakala NP ? | |

## Pacific Forest Province (M2410)

| | | |
|---|---|---|
| Pine-Cypress Forest (9) | Pt. Reyes NS, BLM? | |
| Grand Fir-Douglas Fir Forest (13) | Deschutes NF, Mt. Hood NF?, Williamette NF? | |
| Oregon Oakwoods (22) | BLM | |
| *Coastal Sagebrush (30) | None | California |
| Fescue-Oatgrass (40) | BLM | |
| *Tule Marshes (42) | None | California |

## California Chaparral Province (M2620)

| | | |
|---|---|---|
| *Redwood Forest (6) | None | California |
| Coastal Sagebrush (30) | BLM | |
| *Fescue-Oatgrass (40) | None | California |
| California Steppe (41) | BLM | |
| *Tule Marshes (42) | None | California |

## California Grassland Province (2610)

| | | |
|---|---|---|
| *Juniper-Pinyon Woodland (21) | BLM? | California |
| California Oakwoods (26) | BLM | |
| Saltbush-Greasewood (34) | BLM | |
| California Steppe (41) | BLM | |
| Tule Marshes (42) | Kesterson NWR | |

## Sierran Forest Province (M2610)

| | | |
|---|---|---|
| *Juniper-Pinyon Woodland (21) | BLM?, Inyo NF?, Sequoia NF? | California |
| California Oakwoods (26) | BLM, Shasta NF? | |
| Great Basin Sagebrush (32) | Inyo NF | |
| *California Steppe (41) | BLM? | California |

## Willamette-Puget Forest Province (2410)

| | | |
|---|---|---|
| Cedar-Hemlock-Douglas Fir Forest (2) | BLM | |
| *Oregon Oakwoods (22) | BLM? | Oregon |
| Mosaic of Cedar-Hemlock-Douglas Fir Forest and Oregon Oakwoods (24) | BLM | |

## American Desert Province (3220)

| | | |
|---|---|---|
| ***Mesquite Bosques (23) | None | Arizona |
| Desert: Vegetation largely absent (39) | BLM | |
| *California Steppe (41) | None | California |
| Grama-Tobosa Shrubsteppe (52) | BLM | |

### Palouse Grassland Province (3120)

| | | |
|---|---|---|
| *Cedar-Hemlock-Douglas Fir Forest? (2) | BLM? | Washington |
| Western Ponderosa Forest (10) | BLM | |
| Fescue-Wheatgrass (43) | BLM, Coulee Dam NRA? | |
| Sagebrush Steppe (49) | BLM | |

### Columbia Forest Province (M2110)

| | | |
|---|---|---|
| *Fescue-Wheatgrass (43) | None | Washington, Idaho |
| Wheatgrass-Bluegrass (44) | Nez Perce NF,<br>Payette NF? | |
| *Foothills Prairie (56) | None | Montana |

### Mexican Highlands Shrub Steppe Province (3140)

| | | |
|---|---|---|
| Arizona Pine Forest (18) | Coronado NF | |
| Creosote Bush-Bur Sage (36) | BLM | |
| *Palo Verde-Cactus Shrub (37) | None | Arizona |

### Intermountain Sagebrush Province (3130)

| | | |
|---|---|---|
| Mixed Conifer Forest (5) | Modoc NF | |
| Lodgepole Pine-Subalpine Forest (8) | Toiyabe NF | |
| *Oregon Oakwoods (22) | None | Washington, Oregon |
| Alpine Meadows and Barren (45) | BLM | |

### Upper Gila Mountains Forest Province (M3120)

| | | |
|---|---|---|
| Spruce-Fir-Douglas Fir Forest (19) | Kaibab NF | |
| Southwestern Spruce-Fir Forest (20) | Apache NF | |
| Saltbush-Greasewood (34) | BLM | |
| Creosote Bush-Bur Sage (36) | Tonto NF | |
| Palo Verde - Cactus Shrub (37) | Tonto NF | |
| Alpine Meadows and Barren (45) | Kaibab NF | |

## Colorado Plateau Province (P3130)

| | | |
|---|---|---|
| Western Spruce-Fir Forest (14) | Manti - La Sal NF | |
| Arizona Pine Forest (18) | Zion NP | |
| Mountain Mahogany-Oak Scrub (31) | BLM | |
| Creosote Bush (35) | Grand Canyon NP, Lake Mead NRA? | |
| Grama-Tobosa Shrubsteppe (52) | Grand Canyon NP, Lake Mead NRA? | |

## Rocky Mountain Forest Province (M3110)

| | | |
|---|---|---|
| Saltbush-Greasewood (34) | BLM, Monte Vista NWR? | |
| *Fescue-Wheatgrass (43) | None | Oregon |
| Wheatgrass-Bluegrass (44) | Boise NF, BLM? | |
| Wheatgrass-Needlegrass Shrubsteppe (50) | Shoshone NF | |
| Foothills Prairie (56) | BLM, Helena NF? | |
| Grama-Needlegrass-Wheatgrass (57) | BLM | |

## Wyoming Basin Province (A3140)

| | | |
|---|---|---|
| Douglas Fir Forest (11) | BLM | |
| *Eastern Ponderosa Forest (15) | BLM? | Wyoming |
| Juniper-Pinyon Woodland (21) | BLM | |
| Mountain Mahogany-Oak Scrub (31) | BLM | |
| Saltbush-Greasewood (34) | BLM, Bighorn Canyon NRA? Seedskadee NWR? | |
| Foothills Prairie (56) | BLM, Bighorn NF? | |

## Chihuahuan Desert Province (3210)

| | | |
|---|---|---|
| *Juniper-Pinyon Woodland (21) | BLM? | New Mexico |
| Saltbush-Greasewood (34) | BLM, Guadeloupe Mountains NP? | |
| ***Ceniza Shrub (38) | None | Texas |
| Grama-Tobosa Prairie (48) | Big Bend NP | |
| Shinnery (64) | BLM | |

## Prairie Brushland Province (2520)

| | | |
|---|---|---|
| ***Mesquite - Acacia Savanna (54) | None | Texas |
| ***Mesquite-Live Oak Savanna (55) | None | Texas |
| *Blackland Prairie (68) | None | Texas |
| ***Mesquite-Buffalo Grass (76) | None | Oklahoma, Texas |
| ***Juniper-Oak Savanna (77) | Lyndon B. Johnson NG? | Texas |
| ***Mesquite-Oak Savanna (78) | None | Texas |

## Great Plains Shortgrass Prairie Province (3110)

| | | |
|---|---|---|
| Douglas Fir Forest (11) | Lewis and Clark NF | |
| Western Spruce-Fir Forest (14) | Lewis and Clark NF | |
| *Pine-Douglas Fir Forest (17) | None | Wyoming, Colorado, New Mexic |
| Saltbush-Greasewood (34) | BLM | |
| Sagebrush Steppe (49) | BLM | |
| Foothills Prairie (56) | BLM | |
| *Bluestem-Grama Prairie (62) | None | Oklahoma |
| *Nebraska Sandhills Prairie (67) | None | Wyoming, Nebraska |
| *Cross Timbers (75) | None | Oklahoma |

## Tall-Grass Prairie Province (2530)

| | | |
|---|---|---|
| *Eastern Ponderosa Forest (15) | Fort Niobrara NWR?,<br>Samuel R. McKelvie NF? | Nebraska |
| Wheatgrass-Bluestem-Needlegrass (60) | Des Lac NWR | |
| Sandsage-Bluestem Prairie (63) | Quivira NWR, Great Salt Plains NWR? | |
| *Shinnery (64) | None | Oklahoma |
| ***Bluestem Prairie (66) | Agassiz NWR? | North Dakota, Minnesota, South Dakota, Iowa, Nebraska, Kansas, Oklahoma |
| Cross Timbers (75) | Wichita Mountains NWR | |
| **Maple-Basswood Forest (90) | None | Minnesota |
| *Oak-Hickory Forest (91) | None | Nebraska, Kansas, Iowa |

## Prairie Parkland Province (2510)

| | | |
|---|---|---|
| *Bluestem-Grama Prairie (62) | None | Oklahoma |
| **Bluestem Prairie (66) | None | Oklahoma |
| Blackland Prairie (68) | Lyndon B. Johnson NG | |
| ***Bluestem-Sacahuista Prairie (69) | Aransas NWR? | Texas |
| Mosaic of Bluestem Prairie and Oak-Hickory Forest (73) | Mark Twain NF | |
| **Juniper-Oak Savanna (77) | None | Texas |
| ***Fayette Prairie (79) | None | Texas |
| **Maple-Basswood Forest (90) | None | Illinois |
| *Oak-Hickory Forest (91) | None | Nebraska, Kansas, Oklahoma, Texas, Iowa, Missouri, Illinois, Indiana |
| *Oak-Hickory-Pine Forest (101) | None | Oklahoma, Texas |
| *Southern Mixed Forest (102) | None | Texas |
| Southern Floodplain Forest (103) | Sequoyah NWR, Caddo NG? | |

## Eastern Deciduous Forest Province (2210)

| | | |
|---|---|---|
| **Bluestem Prairie (66) | None | Minnesota, Wisconsin, Illinois, Ohio |
| *Mosaic of Bluestem Prairie and Oak-Hickory Forest (73) | Indiana Dunes NL?, Mark Twain NF? | Illinois, Indiana, Michigan, Ohio, Kentucky, Tennessee, Missouri, Oklahoma |
| Cedar Glades (74) | Mark Twain NF | |
| *Great Lakes Pine Forest (86) | None | Wisconsin, Michigan |
| Northeastern Spruce-Fir Forest (87) | Monongahela NF | |
| Southeastern Spruce-Fir Forest (88) | Great Smoky Mountains NP | |
| *Northern Floodplain Forest (89) | None | Minnesota, Iowa, Wisconsin |
| ***Maple-Basswood Forest (90) | Tamarack NWR? | Minnesota, Iowa, Wisconsin, Illinois |

(continued next page)

| | | |
|---|---|---|
| *Elm-Ash Forest (92) | None | Michigan, Indiana, Ohio |
| *Southern Floodplain Forest (103) | Shawnee NF?, Mark Twain NF? | Indiana, Kentucky, Tennessee, Missouri, Arkansas, Mississippi |

## Laurentian Mixed Forest Province (2110)

| | | |
|---|---|---|
| *Oak Savanna | None | Wisconsin |
| **Maple-Basswood Forest (90) | Chippewa NF? | Minnesota |
| *Oak-Hickory Forest (91) | None | Michigan |
| Elm-Ash Forest (92) | Huron NF | |
| *Beech-Maple Forest (93) | None | Michigan, New York |
| Appalachian Oak Forest (95) | Alleghany NF | |

## Southeastern Mixed Forest Province (2320)

| | | |
|---|---|---|
| ***Blackbelt (80) | None | Mississippi, Alabama |
| Oak-Hickory Forest (91) | Homochitto NF | |
| *Northeastern Oak-Pine Forest (100) | None | New Jersey |
| Southern Mixed Forest (102) | Talladega NF | |

## Outer Coastal Plain Forest Province (2310)

| | | |
|---|---|---|
| **Bluestem-Sacahuista Prairie (69) | None | Louisiana |
| ***Palmetto Prairie (71) | None | Florida |
| *Mosaic of Bluestem Prairie and Oak-Hickory Forest (73) | None | Missouri |
| *Oak-Hickory Forest (91) | None | Missouri, Arkansas |
| *Mangrove (96) | None | Florida |
| *Oak-Hickory-Pine Forest (101) | None | Texas, Louisiana, South Carolina |
| Sand Pine Scrub (105) | Ocala NF, Lake Woodruff NWR? | |

## Everglades Province (4110)

| | | |
|---|---|---|
| **Palmetto Prairie (71) | None | Florida |
| Cypress Savanna (82) | Big Cypress NP | |
| *Southern Mixed Forest (102) | None | Florida |
| *Sand Pine Scrub (105) | None | Florida |
| Subtropical Pine Forest (106) | Big Cypress NP | |

## Puerto Rico Province (M4110)

| | |
|---|---|
| Subtropical Dry Forest (128) | Culebra NWR |
| Subtropical Moist Forest (129) | Caribbean NF |

# APPENDIX F

Distribution among Federally Managed Lands of 91 Major Ecosystem Types in the United States and Puerto Rico that have Minimal Federal Protection (photocopied from Crumpacker, 1979; see p. 29 for definition of minimal and no Federal protection; Table 1 refers to Appendix E).

There are 160 ecosystems in Table 1, out of a total of 315 in the United States and Puerto Rico, which have minimal or no definite Federal protection. They are distributed as follows:

| Land Unit | Managing Agency | Number of Ecosystems with Minimal Federal Protection | |
|---|---|---|---|
| National Forest | FS | 24 | |
| National Grassland | FS | 1 | |
| National Park | NPS | 10 | |
| National Seashore | NPS | 1 | |
| National Wildlife Refuge | FWS | 6 | |
| Public Lands | BLM | 49 | |
| Total | | | 91 |
| Number of ecosystems with no definite Federal protection | | | 69 |
| Grand Total | | | 160 |

All 31 of the Bailey ecoregion provinces on map B are represented in these 160 ecosystems.

# APPENDIX G

States which contain Major Ecosystem Types with No Definite Federal Protection (photocopied from Crumpacker, 1979; see p. 29 for definition of no definite Federal protection and pp. 30 and 52 for meaning of non-trivial ecosystem).

| State | Number of Unprotected Ecosystems which have PNV types that are not protected in any other ecosystem and which are definitely non-trivial ecosystems | which have PNV types that are not protected in any other ecosystem and which might be trivial ecosystems | Number of Other Unprotected Ecosystems | Total Number of Unprotected Ecosystems | State Heritage Program |
|---|---|---|---|---|---|
| Texas | 8 | 1 | 5 | 14 | No |
| Oklahoma | 2 | 1 | 7 | 10 | Yes* |
| California | 0 | 0 | 9 | 9 | Yes*+ |
| Minnesota | 2 | 3 | 1 | 6 | Yes*+ |
| Florida | 1 | 1 | 3 | 5 | No |
| Illinois | 1 | 2 | 2 | 5 | Yes |
| Iowa | 2 | 0 | 3 | 5 | No |
| Michigan | 0 | 0 | 5 | 5 | No |
| Missouri | 0 | 0 | 5 | 5 | No |
| Nebraska | 1 | 0 | 4 | 5 | No |
| Wisconsin | 1 | 1 | 3 | 5 | Yes |
| Indiana | 0 | 0 | 4 | 4 | Yes* |
| Hawaii | 3 | 0 | 0 | 3 | No |
| Kansas | 1 | 0 | 2 | 3 | No |
| Ohio | 0 | 1 | 2 | 3 | Yes* |
| Oregon | 0 | 0 | 3 | 3 | Yes* |
| Washington | 0 | 0 | 3 | 3 | Yes* |
| Wyoming | 0 | 0 | 3 | 3 | Yes* |
| Arizona | 1 | 0 | 1 | 2 | Yes*+ |
| Arkansas | 0 | 0 | 2 | 2 | Yes*+ |
| Kentucky | 0 | 0 | 2 | 2 | Yes* |

| State | Number of Unprotected Ecosystems which have PNV types that are not protected in any other ecosystem and which are definitely non-trivial ecosystems | which have PNV types that are not protected in any other ecosystem and which might be trivial ecosystems | Number of Other Unprotected Ecosystems | Total Number of Unprotected Ecosystems | State Heritage Program |
|---|---|---|---|---|---|
| Louisiana | 0 | 1 | 1 | 2 | No |
| Mississippi | 1 | 0 | 1 | 2 | Yes* |
| New Mexico | 0 | 0 | 2 | 2 | Yes* |
| Tennessee | 0 | 0 | 2 | 2 | Yes* |
| Alabama | 1 | 0 | 0 | 1 | No |
| Colorado | 0 | 0 | 1 | 1 | Yes*† |
| Idaho | 0 | 0 | 1 | 1 | No |
| Montana | 0 | 0 | 1 | 1 | No |
| New Jersey | 0 | 0 | 1 | 1 | No |
| New York | 0 | 0 | 1 | 1 | No |
| North Dakota | 1 | 0 | 0 | 1 | No |
| South Carolina | 0 | 0 | 1 | 1 | Yes* |
| South Dakota | 1 | 0 | 0 | 1 | No |

* Developed or being developed with the formal aid of The Nature Conservancy.

† Started after January 1, 1979.

# APPENDIX H

Major Ecosystem Types of the United States and Puerto Rico for which, according to Davis, 1984, "no agencies appear to have candidate areas[a];" (i.e., no Federal agencies with wilderness responsibilities; photocopied from Davis, 1984, with permission of the author).

Ecoregion[b]
PNV[c]

- 02110 - Laurentian Mixed Forest
  - 092 - Elm Ash Forest
- 00210 - Eastern Deciduous Forest
  - 072 - Oak Savanna
  - 073 - Mosaic Bluestem Prairie and Oak Hickory Forest
  - 089 - Northern Floodplain Forest
  - 090 - Maple Basswood Forest
  - 093 - Beech Maple Forest
- 02310 - Outer Coastal Plain
  - 069 - Bluestem Sacahuista Prairie
  - 071 - Palmetto Prairie
  - 091 - Oak Hickory Forest
- 02320 - Southeastern Mixed Forest
  - 091 - Oak Hickory Forest
  - 100 - Northeastern Oak Pine Forest
  - 102 - Southern Mixed Forest
- 02410 - Willamette Puget Foest
  - 002 - Cedar Hemlock Douglas Fir Forest
  - 024 - Mosaic Oregon Oakwoods and 002
- M2410 - Pacific Forest
  - 022 - Oregon Oakwoods
  - 030 - Coastal Sage
- 02510 - Prairie Parkland
  - 066 - Bluestem Prairie
  - 068 - Blackland Prairie
  - 069 - Bluestem-Sacahuista Prairie
  - 070 - Southern Cordgrass Prairie
  - 073 - Mosaic of B. Prairie & Oak Hkry.
  - 075 - Cross Timber
  - 079 - Fayette Prairie
  - 091 - Oak Hickory Forest
  - 103 - Southern Floodplain Forest
- 02520 - Prairie Brushland
  - 054 - Mesquite Acacia Savanna
  - 055 - Mesquite Live Oak Savanna
  - 076 - Mesquite Buffalo Grass
  - 077 - Juniper Oak Savanna
  - 078 - Mesquite Oak Savanna
- 02530 - Tall Grass Prairie
  - 062 - Bluestem Grama Prairie
  - 063 - Sandsage Bluestem Prairie
  - 066 - Bluestem Prairie
  - 089 - Northern Floodplain Forest
  - 091 - Oak Hickory Forest
- 02610 - California Grassland
  - 034 - Saltbush Greasewood
  - 041 - California Steppe
  - 042 - Tule Marshes
- M2620 - California Chaparral
  - 040 - Fescue Oatgrass
- 03110 - Great Plains Shortgrass Prairie
  - 062 - Bluestem Grama Prairie
  - 063 - Sandsage Bluestem Prairie
  - 064 - Shinnery
- 03120 - Palouse Grassland
  - 043 - Fescue Wheatgrass
- 03210 - Chicuahuan Desert
  - 038 - Caniza Shrub
- 03220 - American Desert
  - 023 - Mesquite Bosques
- 04110 - Everglades
  - 082 - Cypress Savanna
- M4110 - Puerto Rico
  - 128 - Subtropical Dry Forest
- M4220 - Hawaiian Islands
  - 122 - Guava Mixed Forest
  - 124 - Lama Manele Forest
  - 125 - Koa Forest

Total: 50 Ecosystems

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[a] Bureau of Land Management (BLM) inventory incomplete. Table reflects estimates only for BLM lands and assumes BLM lands in Alaska will eventually be considered for wilderness designation, if not, 11 additional ecosystems will be added to this list.

[b] From Bailey (1976).

[c] From Küchler (1966).

# APPENDIX I

Major Ecosystem Types of the United States and Puerto Rico and their Actual or Potential Representation in the National Wilderness Preservation System (photocopied from Davis, 1984, with permission of the author).

| Ecosystem (domain[a]-province[a]-PNV[b]) | Additional units needed (no.) | Remarks (BLM[c] potential excluded) |
|---|---|---|
| I. Polar Domain | | |
| 1210 Arctic Tundra | | |
| 115 - ALDER THICKETS[d] | 2 | |
| 116 - COTTONSEDGE TUNDRA | 1 | |
| 117 - WATERSEDGE TUNDRA | 2 | |
| 118 - DRYAS MEADOWS and BARREN | 2 | |
| M1210 Brooks Range | | |
| 112 - Spruce Birch Forest | 0 | |
| 114 - MUSKEG | 1 | |
| 115 - ALDER THICKETS | 1 | |
| 116 - Cottonsedge Tundra | 0 | |
| 118 - Dryas Meadows and Barren | 0 | |
| 130 - Ice Fields | 0 | |
| 1220 Bering Tundra | | |
| 112 - SPRUCE BIRCH TUNDRA | 2 | |
| 113 - BLACK SPRUCE FOREST | 2 | |
| 115 - ALDER THICKETS | 1 | |
| 116 - Cottonsedge Tundra | 0 | |
| 117 - WATERSEDGE TUNDRA | 2 | |
| 118 - Dryas Meadows and Barren | 0 | |
| 119 - ALEUTIAN MEADOWS | 2 | |
| M1310 Alaska-Aleutian Range | | |
| 112 - Spruce Birch Forest | 0 | |
| 113 - BLACK SPRUCE FOREST | 1 | |
| 115 - Alder Thickets | 0 | |
| 116 - Cottonsedge Tundra | 0 | |
| 117 - WATERSEDGE TUNDRA | 2 | |
| 118 - Dryas Meadows and Barren | 0 | |
| 119 - Aleutian Meadows | 0 | |
| 120 - Aleutian Heath and Barren | 0 | |
| 130 - Ice Fields | 0 | |
| 1320 Yukon Forest | | |
| 112 - Spruce Birch Forest | 0 | |
| 113 - Black Spruce Forest | 0 | |
| 114 - Muskeg | 0 | |
| 115 - ALDER THICKETS | 2 | |
| 116 - COTTONSEDGE TUNDRA | 2 | |
| 117 - WATERSEDGE TUNDRA | 2 | |
| 118 - DRYAS MEADOWS and BARREN | 1 | |
| II. Humid Temperate Domain | | |
| 2110 Laurentian Mixed Forest | | |
| 084 - Great Lakes Spruce Fir Forest | 0 | |
| 085 - Conifer Bog | 0 | |
| 086 - Great Lakes Pine Forest | 0 | |
| 087 - Northeastern Spruce Fir Forest | 0 | |
| 092 - ELM ASH FOREST | 2 | |
| 095 - APPALACHIAN OAK FOREST | 2 | FS[e] potential (Tracy Ridge, Allegheny Islands, PA) |
| 097 - NORTHERN HARDWOODS | 1 | FS potential (Breadloaf, Devil's Den, VT; Kinsman Mountain, NH; Headwaters of the Pine, WI; Sylvania, MI) |
| 098 - Northern Hardwoods Fir Forest | 0 | |
| 099 - Northern Hardwoods Spruce Forest | 0 | |
| M2110 Columbia Forest | | |
| 010 - Western Ponderosa Forest | 0 | |

| Ecosystem (domain[a]-province[a]-PNV[b]) | Additional units needed (no.) | Remarks (BLM[c] potential excluded) |
|---|---|---|
| M2110 Columbia Forest (continued) | | |
| 011 - Douglas Fir Forest | 0 | |
| 012 - CEDAR HEMLOCK PINE FOREST | 1 | NPS[f] potential (Glacier, MT)<br>FS potential (Mallard Larkin, Moose Mountain, Salmo Priest, ID) |
| 013 - Grand Fir Douglas Fir Forest | 0 | |
| 014 - Western Spruce Fir Forest | 0 | |
| 045 - Alpine Meadows and Barren | 0 | |
| 2210 Eastern Deciduous Forest | | |
| 072 - OAK SAVANNA | 2 | |
| 073 - BLUESTEM PRAIRIE and OAK HICKORY MOSAIC | 2 | |
| 074 - Cedar Glades | 0 | |
| 087 - NORTHEASTERN SPRUCE FIR FOREST | 1 | FS potential (Cranberry, WV) |
| 088 - SOUTHEASTERN SPRUCE FIR FOREST | 2 | NPS potential (Great Smokey Mountains, NC, TN) |
| 089 - NORTHERN FLOODPLAIN FOREST | 2 | |
| 090 - MAPLE BASSWOOD FOREST | 2 | |
| 091 - Oak Hickory Forest | 0 | |
| 092 - ELM ASH FOREST | 2 | FS potential (Bald Knob, IL) |
| 093 - BEECH MAPLE FOREST | 2 | |
| 094 - MIXED MESOPHYTIC FOREST | 1 | FS potential (Clifty, KY) |
| 095 - Appalachian Oak Forest | 0 | |
| 097 - NORTHERN HARDWOODS | 1 | NPS potential (Great Smokey Mountains, NC, TN)<br>FS potential (Laurel Fork North, Laurel Fork South, Seneca Creek, WV; Hickory Creek, PA) |
| 100 - NORTHEASTERN OAK PINE FOREST | 1 | |
| 101 - Oak Hickory Pine Forest | 0 | |
| 2310 Outer Coastal Plain Forest | | |
| 069 - BLUESTEM SACAHUISTA PRAIRIE | 2 | |
| 070 - Southern Cordgrass Prairie | 0 | |
| 071 - PALMETTO PRAIRIE | 2 | |
| 081 - Live Oak Sea Oats | 0 | |
| 091 - OAK HICKORY FOREST | 2 | |
| 102 - Southern Mixed Forest | 0 | |
| 103 - Southern Floodplain Forest | 0 | |
| 105 - SAND PINE SCRUB | 2 | FS potential (Juniper Prairie, Buck Lake, Farles Prairie, Alexander Springs, FL) |
| 2320 Southeastern Mixed Forest | | |
| 065 - NORTHERN CORDGRASS PRAIRIE | 1 | FWS[g] potential (Back Bay, VA; Bombay Hook, DE; Brigantine, NJ; Chincoteague, VA) |
| 080 - BLACKBELT | 2 | FWS potential (Noxubee, MS) |
| 081 - Live Oak Sea Oats | 0 | |
| 091 - OAK HICKORY FOREST | 2 | |
| 100 - NORTHEASTERN OAK PINE FOREST | 2 | |
| 101 - Oak Hickory Pine Forest | 0 | |
| 102 - SOUTHERN MIXED FOREST | 2 | |
| 103 - Southern Floodplain Forest | 0 | |
| 104 - POCOSIN | 2 | FS potential (Catfish Lake South, Sheep Ridge, Pond Pine, Pocosin, NC) |
| 2410 Willamette Puget Forest | | |
| 002 - CEDAR HEMLOCK DOUGLAS FIR FOREST | 2 | |
| 024 - CEDAR HEMLOCK DOUGLAS FIR and OREGON OAKWOODS MOSAIC | 2 | |

| Ecosystem (domain[a]-province[a]-PNV[b]) | Additional units needed (no.) | Remarks (BLM[c] potential excluded) |
|---|---|---|
| M2410 Pacific Forest | | |
| 001 - SPRUCE CEDAR HEMLOCK FOREST | 2 | FS potential (Rock Creek, Umpqua Spit, OR)<br>NPS potential (Olympic, WA) |
| 002 - Cedar Hemlock Douglas Fir Forest | 0 | |
| 003 - Silver Fir Douglas Fir Forest | 0 | |
| 004 - Fir Hemlock Forest | 0 | |
| 005 - MIXED CONIFER FOREST | 2 | FS potential (Siskiyou, Trinity Alps, North Fork Smith, CA) |
| 006 - REDWOOD FOREST | 1 | NPS potential (Redwoods, CA) |
| 009 - PINE CYPRESS FOREST | 1 | |
| 014 - Western Spruce Fir Forest | 0 | |
| 022 - OREGON OAKWOODS | 2 | |
| 025 - California Mixed Evergreen Forest | 0 | |
| 030 - COASTAL SAGEBRUSH | 2 | |
| 040 - FESCUE OATGRASS | 1 | |
| 045 - Alpine Meadows and Barren | 0 | |
| 111 - Hemlock Spruce Forest | 0 | |
| 115 - ALDER THICKETS | 2 | FWS potential (Kodiak, AK)<br>FS potential (Red Peak, Malina Bay, AK) |
| 116 - COTTONSEDGE TUNDRA | 2 | FWS potential (Kodiak, AK)<br>FS potential (Chugach, AK) |
| 117 - WATERSEDGE TUNDRA | 2 | FS potential (Sheridan Glacier, Copper River Wetlands, Bering Lake, AK) |
| 118 - Dryas Meadows and Barren | 0 | |
| 130 - Ice Fields | 0 | |
| 2510 Prairie Parkland | | |
| 066 - BLUESTEM PRAIRIE | 2 | |
| 068 - BLACKLAND PRAIRIE | 2 | |
| 069 - BLUESTEM SACAHUISTA PRAIRIE | 2 | |
| 070 - SOUTHERN CORDGRASS PRAIRIE | 2 | |
| 073 - BLUESTEM PRAIRIE and OAK HICKORY MOSAIC | 2 | |
| 075 - CROSS TIMBER | 2 | |
| 079 - FAYETTE PRAIRIE | 2 | |
| 091 - OAK HICKORY FOREST | 2 | |
| 103 - SOUTHERN FLOODPLAIN FOREST | 2 | |
| 2520 Prairie Brushland | | |
| 054 - MESQUITE ACACIA SAVANNA | 2 | |
| 055 - MESQUITE LIVE OAK SAVANNA | 2 | |
| 076 - MESQUITE BUFFALO GRASS | 2 | |
| 077 - JUNIPER OAK SAVANNA | 2 | |
| 078 - MESQUITE OAK SAVANNA | 2 | |
| 2530 Tall Grass Prairie | | |
| 060 - WHEATGRASS BLUESTEM NEEDLEGRASS | 1 | |
| 062 - BLUESTEM GRAMA PRAIRIE | 2 | |
| 063 - SANDSAGE BLUESTEM PRAIRIE | 2 | |
| 066 - BLUESTEM PRAIRIE | 2 | |
| 067 - NEBRASKA SANDHILLS PRAIRIE | 2 | FWS potential (Crescent Lake, Valentine, NB) |
| 089 - NORTHERN FLOODPLAIN FOREST | 2 | |
| 091 - OAK HICKORY FOREST | 2 | |
| 2610 California Grassland | | |
| 034 - SALTBUSH GREASEWOOD | 2 | |
| 041 - CALIFORNIA STEPPE | 2 | |
| 042 - TULE MARSHES | 2 | |

| Ecosystem (domain[a]-province[a]-PNV[b]) | Additional units needed (no.) | Remarks (BLM[c] potential excluded) |
|---|---|---|
| M2610 Sierran Forest | | |
| 005 - Mixed Conifer Forest | 0 | |
| 007 - Red Fir Forest | 0 | |
| 010 - WESTERN PONDEROSA FOREST | 2 | FS potential (Cinder Butte, Devil's Garden, Lost Creek, Timbered Crater, CA) |
| 021 - JUNIPER PINYON WOODLAND | 2 | |
| 025 - CALIFORNIA MIXED EVERGREEN FOREST | 2 | FS potential (Trinity Alps, (CA) |
| 026 - CALIFORNIA OAKWOODS | 2 | FS potential (Tuolomne River, Moses, Ishi, Oat Mountain, CA) |
| 029 - Chaparral | 0 | |
| 045 - Alpine Meadows and Barren | 0 | |
| M2620 California Chaparral | | |
| 005 - Mixed Conifer Forest | 0 | |
| 021 - JUNIPER PINYON WOODLAND | 1 | FS potential (Granite Peak, Quatal, Sawmill-Badlands, Fox Mountain, Antimony, Cuyama, Sespe-Frazier, CA) |
| 026 - California Oakwoods | 0 | |
| 029 - Chaparral | 0 | |
| 030 - COASTAL SAGEBRUSH | 1 | |
| 040 - FESCUE OATGRASS | 2 | |
| 041 - CALIFORNIA STEPPE | 2 | |
| III. Dry Domain | | |
| 3110 Great Plains Shortgrass Prairie | | |
| 015 - EASTERN PONDEROSA PINE | 2 | FS potential (Tongue River, Breaks, MT; Pine Ridge, NB) |
| 016 - BLACK HILLS PINE FOREST | 1 | NPS potential (Wind Cave, SD) |
| 021 - JUNIPER PINYON WOODLAND | 2 | FS potential (Canadian River, NM) foreclosed by PL 96-550 |
| 049 - SAGEBRUSH STEPPE | 2 | |
| 056 - FOOTHILLS PRAIRIE | 2 | |
| 057 - GRAMA NEEDLEGRASS WHEATGRASS | 1 | FWS potential (Charles Russell, MT)<br>FS potential (Laramie Peak, WY) |
| 058 - GRAMA BUFFALO GRASS | 2 | FS potential (Willow Creek, CO) foreclosed by PL 96-560 |
| 059 - Wheatgrass Needlegrass | 0 | |
| 060 - WHEATGRASS BLUESTEM NEEDLEGRASS | 1 | |
| 061 - WHEATGRASS GRAMA BUFFALO GRASS | 1 | |
| 062 - BLUESTEM GRAMA PRAIRIE | 2 | |
| 063 - SANDSAGE BLUESTEM PRAIRIE | 2 | |
| 064 - SHINNERY | 2 | |
| 089 - NORTHERN FLOODPLAIN FOREST | 1 | |
| M3110 Rocky Mountain Forest | | |
| 010 - Western Ponderosa Forest | 0 | |
| 011 - Douglas Fir Forest | 0 | |
| 013 - Grand Fir Douglas Fir Forest | 0 | |
| 014 - Western Spruce Fir Forest | 0 | |
| 017 - Pine Douglas Fir Forest | 0 | |
| 018 - ARIZONA PINE FOREST | 2 | FS potential (Ashdown Gorge, UT) |
| 019 - SPRUCE FIR DOUGLAS FIR FOREST | 2 | FS potential (Fishlake Mountain, UT)<br>NPS potential (Cedar Breaks, UT) |
| 020 - Southwestern Spruce Fir Forest | 0 | |

| Ecosystem (domain[a]-province[a]-PNV[b]) | Additional units needed (no.) | Remarks (BLM[c] potential excluded) |
|---|---|---|
| M3110 Rocky Mountain Forest (continued) | | |
| 021 - JUNIPER PINYON WOODLAND | 1 | NPS potential (Bryce Canyon, Zion, UT, Dinosaur, CO, UT)<br>FS potential (Ashdown Gorge, UT) |
| 031 - MOUNTAIN MAHOGANY OAK SCRUB | 1 | NPS potential (Dinosaur, CO, UT)<br>FS potential (Wellsville Mtn., Nephi, Birdseye, Santaquin, UT) |
| 032 - GREAT BASIN SAGEBRUSH | 1 | FS potential (Dromedary, UT) |
| 034 - SALTBUSH GREASEWOOD | 1 | FS potential (Nephi, Fishlake Mountain, UT) |
| 045 - Alpine Meadows and Barren | 0 | |
| 046 - FESCUE MOUNTAIN MUHLY PRAIRIE | 2 | FS potential (Sangre de Cristo, CO) |
| 049 - SAGEBRUSH STEPPE | 1 | NPS potential (Dinosaur, CO, UT; Grand Teton, WY)<br>FS potential (Palisades, Cloud Peak, WY; Wellsville Mt. Jefferson, MT) |
| 056 - FOOTHILLS PRAIRIE | 1 | FS potential (Snowies, MT) |
| 057 - GRAMA NEEDLEGRASS WHEATGRASS | 2 | |
| 3120 Palouse Grassland | | |
| 043 - FESCUE WHEATGRASS | 2 | |
| 044 - WHEATGRASS BLUEGRASS | 2 | FS potential (Imnaha Face, Lords Flat-Somers Pt, Snake River, Buckhorn, Mountain Sheep, Cook Ridge, Wildhorse, OR) |
| M3120 Upper Gila Mountains Forest | | |
| 017 - Pine Douglas Fir Forest | 0 | |
| 018 - ARIZONA PINE FOREST | 1 | FS potential (San Francisco Peak, Castle Creek, Juniper Mesa, Red Rock-Secret Mountain, Wet Beaver, Fossil Springs, AZ) |
| 021 - Juniper Pinyon Woodland | 0 | |
| 028 - Oak Juniper Woodland and Mountain Mahogany Mosaic | 0 | |
| 047 - GRAMA GALLETA STEPPE | 1 | FS potential (adjacent to Apache Kid, NM) foreclosed by PL 96-550 |
| 052 - GRAMA TOBOSA SHRUBSTEPPE | 1 | |
| 3130 Intermountain Sagebrush | | |
| 010 - WESTERN PONDEROSA FOREST | 1 | FWS potential (Desert, NV)<br>FS potential (Woodpecker, CA; Highland Ridge, Grant Range, Ruby Mountains, Quinn, Wheeler Peak, Mt. Moriah, NV) |
| 021 - JUNIPER PINYON WOODLAND | 1 | FWS potential (Desert, NV)<br>FS potential (Highland Ridge, Ruby Mountains, NV; Sweetwater, Excelsior, Deep Wells, Blanco Mountain, Birch Creek, Black Canyon, Paiute/Mazourka, Boundary Peak/White Mountain, Andrews Mountain, Scodies, Sugarloaf, CA) |
| 031 - MOUNTAIN MAHOGANY OAK SCRUB | 2 | FS potential (Arc Dome, NV) |
| 032 - GREAT BASIN SAGEBRUSH | 1 | FWS potential (Desert, Sheldon, NV; Hart, Malheur, OR)<br>FS potential (Sweetwater, Deep Wells, Excelsior, Blanco Mountain, Birch Creek, Black Canyon, Tinnemaha, Grant Lake, CA; Grant Range, Quinn, NV) |

| Ecosystem (domain[a]-province[a]-PNV[b]) | Additional units needed (no.) | Remarks (BLM[c] potential excluded) |
|---|---|---|
| 3130 Intermountain Sagebrush (continued) | | |
| 034 - Saltbush Greasewood | 0 | |
| 039 - DESERT | 1 | |
| 042 - TULE MARSHES | 2 | |
| 044 - WHEATGRASS BLUEGRASS | 2 | NPS potential (Zion, UT) |
| 049 - Sagebrush Steppe | 0 | |
| P3130 Colorado Plateau | | |
| 017 - Pine Douglas Fir Forest | 0 | |
| 020 - Southwestern Spruce Fir Forest | 0 | |
| 021 - Juniper Pinyon Woodland | 0 | |
| 032 - GREAT BASIN SAGEBRUSH | 2 | FS potential (Kanab Creek, AZ) |
| 033 - BLACKBRUSH | 2 | NPS potential (Arches, Canyonlands, UT) |
| 034 - SALTBUSH GREASEWOOD | 2 | NPS potential (Arches, Capitol Reef, UT; Colorado, CO) |
| 047 - Grama Galleta Steppe | 0 | |
| 051 - GALLETA THREE AWN SHRUBSTEPPE | 2 | NPS potential (Canyonlands, Capitol Reef, UT) |
| 052 - GRAMA TOBOSA SHRUBSTEPPE | 2 | |
| 3140 Mexican Highlands Shrub Steppe | | |
| 018 - Arizona Pine Forest | 0 | |
| 027 - Oak Juniper Woodland | 0 | |
| 036 - CREOSOTE BUSH BUR SAGE | 2 | FS potential (Rincon Mountains, AZ) |
| 052 - GRAMA TOBOSA SHRUBSTEPPE | 2 | FS potential (Hell Hole, NM) |
| A3140 Wyoming Basin | | |
| 034 - SALTBUSH GREASEWOOD | 2 | |
| 049 - SAGEBRUSH STEPPE | 2 | |
| 050 - WHEATGRASS NEEDLEGRASS SHRUBSTEPPE | 2 | |
| 057 - GRAMA NEEDLEGRASS WHEATGRASS | 2 | |
| 3210 Chihuahuan Desert | | |
| 027 - OAK JUNIPER WOODLAND | 2 | FS potential (Guadalupe Escarpment, NM) |
| 034 - SALTBUSH GREASEWOOD | 2 | |
| 038 - CENIZA SHRUB | 2 | |
| 048 - GRAMA TOBOSA PRAIRIE | 2 | NPS potential (Big Bend, TX) |
| 052 - Grama Tobosa Shrubsteppe | 0 | |
| 053 - Trans Pecos Shrub Savanna | 0 | |
| 064 - SHINNERY | 2 | |
| 3220 American Desert | | |
| 021 - JUNIPER PINYON WOODLAND | 1 | NPS potential (Death Valley, CA)<br>FWS potential (Desert, NV) |
| 023 - MESQUITE BOSQUES | 2 | |
| 034 - SALTBUSH GREASEWOOD | 2 | NPS potential (Death Valley, CA) |
| 035 - CREOSOTE BUSH | 1 | NPS potential (Death Valley, CA)<br>FWS potential (Desert, NV) |
| 036 - Creosote Bush Bur Sage | 0 | |
| 037 - Palo Verde Cactus Shrub | 0 | |
| 039 - DESERT | 2 | |
| IV. Humid Tropical Domain | | |
| 4110 Everglades | | |
| 082 - CYPRESS SAVANNA | 2 | |
| 083 - EVERGLADES | 1 | |
| 096 - Mangrove | 0 | |
| 106 - SUBTROPICAL PINE FOREST | 1 | |
| M4110 Puerto Rico | | |
| 128 - SUBTROPICAL DRY FOREST | 2 | |
| 129 - SUBTROPICAL MOIST FOREST | 2 | FS potential (El Cacique, PR) |

| Ecosystem (domain[a]-province[a]-PNV[b]) | Additional units needed (no.) | Remarks (BLM[c] potential excluded) |
|---|---|---|
| M4220 Hawaiian Islands | | |
| 121 - Sclerophyllous Forest Shrub and Grass | 0 | |
| 122 - GUAVA MIXED FOREST | 2 | |
| 123 - OHIA LEHUA FOREST | 1 | |
| 124 - LAMA MANELE FOREST | 2 | |
| 125 - KOA FOREST | 2 | |
| 126 - KOA MAMANI PARKLAND | 1 | |
| 127 - Grassland Microphyllous Shrub and Barren | 0 | |

[a] Domain and province as described by Bailey (1976).

[b] PNV = potential natural vegetation (Küchler 1966).

[c] BLM = Bureau of Land Management.

[d] Capitalized PNV indicates that the ecosystem is not adequately represented in either the NWPS or in the two equivalent state systems (California and New York). Adequate representation = two or more representative areas of at least 1000 acres each.

[e] FS = USDA Forest Service.

[f] NPS = National Park Service.

[g] FWS = U.S. Fish and Wildlife Service.

Contract Paper No. 2

# Geographic Information Systems as A Tool to Assist in the Preservation of Biological Diversity

Duane Marble
SUNY - Buffalo

December 1985

Office of Technology Assessment
U.S. Congress

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# GEOGRAPHIC INFORMATION SYSTEMS AS A TOOL TO ASSIST IN THE PRESERVATION OF BIOLOGICAL DIVERSITY

A Report Prepared for the

Office of Technology Assessment
Congress of the United States

Contract No.: 533-4735.0

by

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December 1985

# Geographic Information Systems as a Tool to Assist in the Preservation of Biological Diversity

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## Abstract

Maintaining the continuing biological diversity of plants and animals is a large, complex task which requires the acquisition, storage, retrieval and manipulation of massive quantities of spatial data. Data relating to in situ biological resources is mainly spatial in nature, that is each entity, or group of entities, in the database has associated with it a coordinate-based measure of location. Traditional approaches to spatial data management have emphasized manual cartographic methods for data storage and manipulation, but these are being increasingly replaced by approaches based upon modern computer technology.

Standard data management approaches, as implemented in database management systems, become highly inefficient when applied to many forms of spatial data. Special approaches are needed and these are contained in what are called **geographic information systems**. Geographic information systems are being utilized today by many governmental and private organizations for the management of spatial data and, in a number of cases, these data are biological in nature.

Creation of a viable geographic information system and its associated spatial database is a high technology task requiring substantial, systematic activities in the areas of planning for and analysis of data acquisition and utilization. Many of the databases for natural systems which have been and are being constructed do not closely follow the methodologies developed for effective database design. As a result, both the short-run and long-run utility of these databases, many of which are being created by Federal agencies, is significantly diminished.

The input data streams for biologically-oriented, geographic information systems reflect a large component of manual, field-based observation. While there is a growing contribution from instrument-based approaches (e.g. remote sensing) to data collection, these tend to be constrained to very specific classes of biological entities and spatial scales. They do, however, often provide data which cannot be obtained by conventional methods.

Both the current availability and utility of databases relating to in situ biological systems are severely limited by the high degree of fragmentation and lack of common structural definitions which arise from current institutional approaches to the creation of these databases. Some short-run improvements in the situation may be generated through changes in the institutional approaches

which will permit greater emphasis upon proper spatial database design and implementation.

Substantial long-run and medium-run improvements can be made in the present situation through coordinated development of spatial database indexing schemes, encouraging substantial new research and development activities in the areas of geographic information systems and data acquisition methodologies, and by improving the existing level of understanding of the value of structured database design methodologies.

# Geographic Information Systems as a Tool to Assist in the Preservation of Biological Diversity

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# Geographic Information Systems as a Tool to Assist in the Preservation of Biological Diversity

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## Introduction

Maintaining the continuing biological diversity of plants and animals is a massive and complex task which is now recognized as being of significant national importance. The nation's interest in this area arises, in part, out of a desire to keep open future options in such areas as agriculture, medical research and genetic engineering. To carry out this task in an efficient and viable fashion requires the acquisition, storage, retrieval, manipulation and display of massive quantities of biological data which, for in situ observations, must be explicitly located in both space and time. This space-time database can provide the necessary basis for both planning and monitoring the results of activities designed to maintain biological diversity. Needless to say, it represents a planning and management tool and cannot by itself define operational solutions to the problem.

The present report examines the status of existing technology for data capture, storage, retrieval and manipulation of space-time data relating to in situ biological resources. The process of storage and manipulation of spatial databases has a long tradition based upon the manual processing of analog map docuements, but today it is increasingly based upon modern computer technology. The specialized computer systems which have been developed to handle large spatial databases are known as **geographic information systems** (GIS) and are widely used in the operational handling of large volumes of spatial data (e.g., in forest management, land-use planning, facility management of public utility systems, etc., - see several examples in Marble, Calkins & Peuquet, 1984).

As with many computer-based systems, the advent of the GIS has not only increased the efficiency of existing, manual map processing but has also enabled us to extend our operations to levels which were previously impossible to reach because of the sheer volume of data to be handled. An understanding of current GIS structures and operations may be most easily reached by contrasting them with traditional manual approaches to spatial data handling.

## An Example of Traditional Data Management

An interesting example of traditional map-based approaches to the storage and retrieval of spatially-oriented, biological data may be found in Fitter (1978). His Atlas of the Wild Flowers of Britain and Northern Europe shows the spatial distribution of nearly 2000 wild plants in Europe between the Alps and Iceland. The spatial data stored in the Atlas are highly generalized since the individual maps measure only about 1 1/2" by 2 1/2" for the entire region covered. In the "Introduction" to the Atlas Fitter notes that a more detailed data collection and storage operation (the Atlas Florae Europaeae) has been underway for several years, but that it will be "many years" before it is complete.

Data sources for creation of the database included Fitter´s own field observations, published maps, and information contained in scientific articles in a number of different journals. Fitter also notes that the quality of the data contained in the Atlas varies significantly from region to region. [No attempt is made to provide specific information on the spatial variations in data quality - a failing common to nearly all map-based data storage systems.] An attempt is made, through the presentation of supplemental maps on seasonal temperatures, rainfall, etc., to permit visual correlations to be made by the user between the mapped distributions and influential environmental factors. (This is a major factor in the operation of many GIS, with some organizations having nearly a hundred different data "layers" available for cross-correlation analysis.) Data pertaining to the plants themselves, as opposed to data relating to their spatial distribution, are stored in Balmey, Fitter and Fitter (1978). Because of the extreme difficulty in obtaining the data (a difficulty which, as we shall see, poses a continuing, major problem with respect to biological data) no explicit temporal data is included.

While a substantial amount of work went into the creation of Fitter's database, its ultimate utility is reduced by the overall lack of detail and by the format which restricts users to qualitative, visual comparisons of spatial factors. It would be very difficult to pose a query to this database regarding, say, the number of species to be found within a specific subarea since obtaining an answer would require manual processing of the entire database.

Could a more useful, quantitative database be constructed using modern computer technology? The answer is clearly yes, but a number of critical questions need to be examined first. For instance, a greater level of detail than presented by Fitter is certainly desirable, but how much detail is either necessary or possible? What is necessary is defined upon the basis of the problems which must be solved; what is possible is often a joint function of the current level of technology, the resources available and the institutional infrastructure within which the activity takes place.

If detail is to be somehow reduced, how is this to be done without significant loss of utility? How is internal consistency in the database to be achieved and maintained? These, and many other related questions, must be answered as part of the logical design of any database system. A simple example will assist in pointing out the critical role which structural database design must play in answering these, and other critical questions, which must be addressed in the creation of viable databases directed toward the preservation of in situ biological diversity.

## A Simplistic, Large Biological Database

Biological entities, viewed in situ, exhibit a large number of individual attributes. An entity in the database may be viewed as an individual plant or animal which is described in terms of a series of attributes which refer to the its spatial location at a specific point in time, its age, size, genetic structure, etc. We could, on a very simplistic and conceptual level, view all in situ biological entities as being described by entries in a very large table where the individual entities are represented by the rows and their various attributes by the columns. (It is not necessary to assume that every attribute has a valid value for each entity, a "not appropriate" code may substitute for a valid observation.) This approach represents a simple form of the relational database model and the use here is of a single, "universal relation" as discussed in Ullman (1982).

It is interesting to speculate if biology is currently capable of defining a stable list of attributes for this hypothetical table. If so, perhaps this simple model, and its logical components, represents a closer approach to the "omnifarious data bank for biology" which is discussed in a somewhat confused fashion in a National Research Council report (1985) and by Holden (1985).

Within some arbitrary period of time, a day or a week or a year, many changes will take place in our hypothetical table. Some entities will die (deletion of rows), others will be born (addition of rows), and many of the attribute values will be subject to change. A discrete space-time model of all in situ biological data might be conceptualized as a series of these tables, each of which would represent the characteristics of the biologic population of our planet during a small interval of time. If these discrete time samples are taken close enough together to be viewed as approximating a continuous model, we then have a database containing the "world lines" of all biological entities from birth to death. Obviously, this "no sparrow shall fall" database would not only be far beyond our capabilities to create and maintain, it would also contain very substantial volumes of data which would be of little or no utility to anyone. Never the less, it does represent a simple conceptual model of a master biological database oriented toward in situ data.

## Pragmatic Structuring of the Database

To be of practical utility, our simple, conceptual database must be reduced in size by many orders of magnitude. This may be accomplished thru actions such as increasing the length of the time periods between observations, reducing the total number of attributes which are included, limiting the database to areas of the planetary surface which are of major interest, redefining the entities on an aggregative basis, or selectively eliminating entities upon the basis of some external criteria of relevancy. In reality, because of the massive data volumes involved, all of these approaches must be implemented in order to bring the database down to a viable size.

However, implementation of any of these approaches outside the context of a general, structured spatial database design will lead only to substantial decreases in the utility of the database and increase the chance of ultimate failure (Lucas, 1975; Tomlinson, Calkins & Marble, 1976; Teorey & Fry, 1982).

In the present context, where maintenance of biological diversity and the associated reservoirs of germ plasm is of paramount interest, species abundance could possibly serve as one basic screening device to reduce the volume of data. Figure 1 illustrates the situation posited in the simple model above -- a more or less constant level of information about each entity. Figure 2 shows an approach much more in accord with reality where certain minimal amounts of information are available about all species, but the information level is raised to significantly higher levels for species whose occurrence in nature is very limited.

Since many species are also of significant commercial importance and are, as a result, both quite common and quite well-known because of our special interest in them (the reverse of what is shown in Figure 2), we would expect to see - in reality - a U-shaped curve of information levels where the least information is available on species of moderate abundance

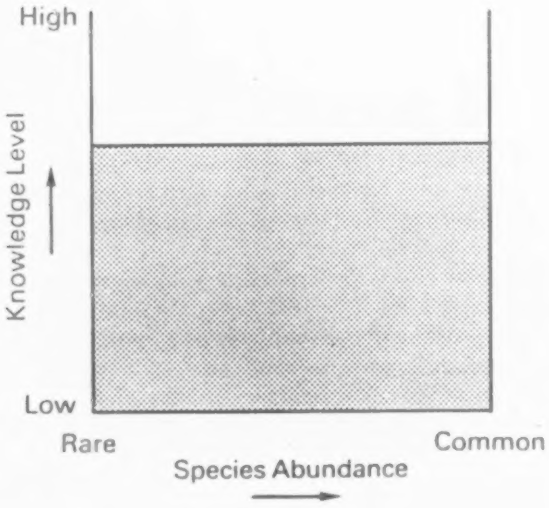

Figure 1

Uniform Information Levels

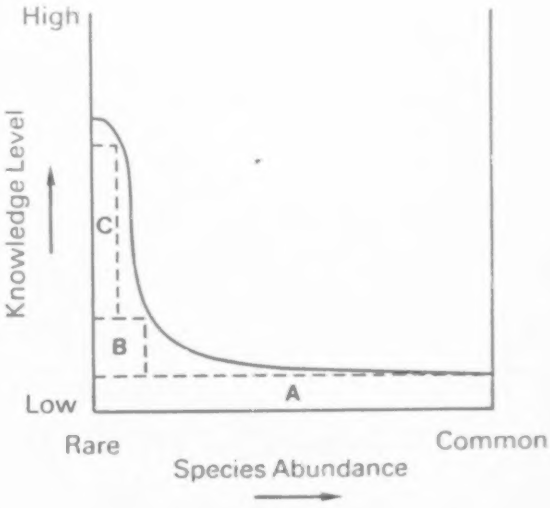

Figure 2

Structured Information Levels

Even with rigorous constraints imposed upon the size and structure of the simple, conceptual model many operational questions remain to be answered. How are the data to be managed so as to be as widely useful as possible? What techniques can be used to capture the vast quantities of data still called for? As we will see, answers to these and related questions involve a complex mix of technical and institutional aspects and cannot be completely resolved at the present time.

## Computer-Based Tools for Data Management

In recent years a great deal of attention has been given to the development and application of computer-based tools for data management to assist in the efficient utilization of the large databases which have developed in many fields. These tools are often generic in nature in that they may be applied, without physical modification, to a wide range of data management applications.

### The Database Management System (DBMS)

Complex software packages to aid the manager and researcher have been developed over the last three decades, and are known as **database management systems** (DBMS´s). These are widely used in many different areas of application and over a wide range of computer hardware (Date, 1975; Date, 1983; Martin, 1976; and Ullman, 1982). Small, and in many cases efficient, systems are available for personal computers but databases of any significant size require substantial amounts of mass storage and are generally found on medium to large machines (the large minicomputer and mainframe environment). [This distinction is becoming blurred by the increasing power and speed of microprocessors. Machines such as the MicroVAX II are available with substantial mass storage for under $50,000 and even the IBM PC/AT may be set up with mass storage systems exceeding 100MB in size.]

The concept of **data independence** which is built into nearly all modern DBMS permits logically identical databases to be maintained on different types of computer hardware using versions of the DBMS adapted to each computer environment. Thus, the applications programs using the DBMS are not forced to utilize different interfaces when the computer environment changes. Physically, the DBMS consists of a computer environment (hardware plus the associated operating system) and an associated specialized software package. Too often the view of both users and managers is restricted to just the physical components of the DBMS and its most critical structural component (the logical design of the database) is neglected.

The DBMS requires this **conceptual model** to efficiently organize its contents; common, non-spatial, logical models include hierarchical, network and relational implementations. Of these the relational model (which views the data as a series of tables similar to the simplistic biological example outlined above) is the most flexible, but it currently lacks the power to handle very large numbers of entities (much above 100K records) without significant losses in efficiency. Existing DBMS systems usually implement only one of these conceptual models which deal with the nature of the **relationships** which exist between entities and groups of entities (as well as with the specification of the entities) within the database (Teorey and Fry, 1982).

It is critical to realize that a conceptual and logical structure must be present before a DBMS may be effectively constructed and utilized. A major bridge may not be constructed without detailed plans, and a database system cannot be brought into effective existence without a consistent conceptual and logical model of its structure. Attempts to build a database, or a logically linked series of similiar databases, to support the maintainence of biological diversity without first establishing a viable conceptual structure is likely to lead only to ultimate failure.

Another important factor in effective use of DBMS technology lies in consistent and commonly accepted definitions of the entities, attributes, and relationships within the database. These definitions exist within the context of a **data dictionary** which must be created prior to the use of the DBMS to create the database. Without both initial and continuing adherence to the data dictionary, the database becomes flawed and relatively useless. To be effective, the initial creation of the data dictionary (as well as its continued maintenance) must involve inputs from significant elements of the user community and, as in the day-to-day operation of the DBMS, these tasks are normally supervised by an individual who is known as the **data base administrator** (DBA). A number of Federal agencies (such as the Department of Agriculture) have appointed departmental-level DBA´s, while others operate at the Agency level (e.g, the USGS).

The design and creation of a large, modern database is a complex activity which frequently has significant impacts upon methods of data collection, utilization, and administration. Despite the power of modern database technology, the standard DBMS is not capable of efficiently handling large quantities of complex space-time data. Specialized data management and manipulation systems have been created to fill this gap.

## Spatial Data Management: The Geographic Information System (GIS)

The majority of the DBMS available today were developed for non-spatial applications (situations where location in space and time do not matter) and they encounter substantial difficulties when space-time dependent data and spatial relationships must be accommodated. Special approaches need to be adopted in situations where explicit consideration of these parameters is required, as is the case in nearly all databases dealing with in situ natural systems. The special data management systems which have been developed to handle explicit spatial data are known as **geographic information systems** (GIS) and are widely used in areas such as land-use planning, forest management, oceanography, etc.

The problems encountered by standard DBMS structures when confronted by time and space-dependent data sets arise out of their inability to efficiently handle inquiries which involve spatial relationships between entities within the database. (E.g., find all bodies of water greater than five hectares in size which lie within one and one-half kilometers of a forest road network.) The GIS has evolved to solve problems of this type and often does so through a combination of special, spatial data handling software and standard DBMS technology. For example, a standard DBMS may be utilized to handle the non-spatial attributes while a closely linked spatial data handling system deals with those queries which are spatial in nature. This is the approach used in ARC/INFO, one of the most widely used, turn-key, geographic information systems: INFO is a commercial DBMS while ARC is a specially constructed spatial data handling system.

### Some Examples of Biological GIS

In the past several years, GIS technology has been widely adopted in a number of organizations (both governmental and private) concerned with natural systems. The following examples provide some indication of the manner in which GIS technology is currently being applied in a biological contex

### U. S. Fish and Wildlife Service

An example of the biological application of a GIS may be seen in Asherin, Short and Roelle (1979). The U.S. Fish and Wildlife Service had an urgent need to evaluate the impact of Western coal development upon wildlife habitats so that unique, high quality habitats of special importance could be identified and protected. To assist in the solution of this problem a spatial database was created for a portion of southeastern Montana. Data sets contained in the GIS included, among others, digital maps of land use and surface cover, and data on feeding and breeding locations for 235 species of birds, 74 species of mammals, 14 species of reptiles and 6 species of amphibians.

The digital surface cover information was manipulated by the GIS to obtain a number of habitat parameters of ecological interest and the species database was used to create faunal diversity and juxtaposition indices. A statistical analysis was performed to determine the most efficient predictor of wildlife habitat quality and this variable was then mapped for the study area and related to areas of potential coal development. Nearly two hundred similar investigations have been carried out by the Fish and Wildlife Service (Asherin, personal communication, 1985).

### The Nature Conservancy

Another, and much less spatially restricted, example may be found in the database systems established by the Nature Conservancy. The Natural Heritage Program is concerned with the preservation of biological diversity and is implemented through a series of state-level agreements which call for the establishment of local databases which are constructed according to a set of consistent, national standards (Nature Conservancy, 1982). Local implementation of these standards reflects the level of local resources, so the technology employed may vary from highly automated to manual. On a national level only the element occurrence records (EOR), which show occurrences in the landscape of elements defined in the master classification system, are consistently computerized.

The classification system used is a multi-level one which reflects both the very large number of species and the directed interest in less abundant species. The initial, and broadest level, of classification is the community type wherein each type has a species complement which is more or less characteristic of it. By preserving examples of all community types found in an area, the Conservancy estimates that 85-90 percent of the biological diversity in the area will be preserved.

The remaining 10-15 percent of the flora and fauna species must be treated on an individual basis and here the system is called upon to maintain species-level records. In cases where the level of occurrence is very low, records may be maintained on an individual basis. This hierarchical structuring represents an implementation of the structured information levels shown in Figure 2.

Element occurrences represent the basic building block of the system and are available in digital form. Each EOR is a "locational record representing a single extant habitat which sustains or otherwise contributes to the survival of a population or self-sustaining example of a particular element" (Nature Conservancy, 1982, p. 6.1-1). At the national level EOR records are not currently maintained within the context of a modern GIS due to lack of resources, although the possible application of microprocessor-based GIS is currently being examined. At the state level, a number of the State Heritage Programs make use of GIS technology but this is on an "as available" basis if access can be obtained to a GIS operated by one of the state agencies.

One operational purpose of the National Heritage System is to identify elements of the biological community in terms of their frequency of occurrence and to focus operational attention upon those species which are relatively rare. This is accomplished through an element ranking operation within the data system. Of the systems identified during the course of this investigation, the system maintained by the Nature Conservancy would appear to come closest to meeting the anticipated goals of a national GIS for maintenance of biological diversity. The current operations of the system are severely restricted by the interaction between the broad national scope of the desired coverage and the limited resources available.

For some additional biological GIS examples, see Brigham and Treworgy (1985), Anonymous (1984), Parker and McConaghy (1983), VPI & SU (1984), and DuBrock, et al, (1981). The investigations reported upon in the BLM's Piceance Basin Resource Management Plan and Environmental Impact Statement, U. S. Department of the Interior (1984), were carried out using a completely digital database within the context of a GIS. Numerous other Bureau of Land Management GIS studies are briefly noted in Appendix A.

## The Structure of a GIS

A GIS consists of four operational subsystems:

> A **data input subsystem** which deals with digital data capture from map documents and entry of these data elements, as well as other externally generated space-time data (e.g., from remote sensing systems), into the GIS. Data sets within the GIS are often viewed as a series of **layers**, each representing the spatial extent of a major data element (e.g., soils, vegetation types, etc.).

A **data storage and retrieval subsystem** which is responsible for storing the geographic data in an efficient fashion for subsequent search and retrieval. This frequently involves the use of a regular or irregular subdivision of the region with, in some cases, a hierarchical structure imposed upon the sets of spatial units.

A **data manipulation subsystem** which performs a variety of computational operations (e.g. corridor definition or overlay) upon the material stored in the database. (See, for example, Dangermond (1982).) Primary output from this subsystem, as well as output from associated modelling systems, is often returned to the data storage and retrieval subsystem as new data layers.

A **cartographic data output subsystem** which creates map displays of materials contained in the database or created by the data manipulation subsystem or by any external models which may be linked to the GIS. These displays are complex and difficult to create in an effective fashion; however their utility as a "window" upon the space-time database is unrivaled.

## Special Problems of Spatial Data Handling

Certain unique problems are encountered in the handling of large volumes of spatial data. These include the utilization of different data models and the generalization of the data with regard to the spatial dimension of the database.

## Spatial Data Models

Every data model represents an abstraction of a real-world situation; some factors are ignored and others modified. Usually there are a number of different data models available to the investigator and, for each of these, a variety of data structure implementations are possible (see Figure 3). Two basic data models have characterized most of the GIS developed to date: regular tessalations and direct boundary representation. In the former case regular subdivisions of the region are established (e.g., grid cells) and data relating to the entities found within each grid cell is recorded. (This is a common method of recording information on avian habitats where a cell will be given a specific value, e.g., five, if a nest is observed, a three if feeding is observed and a one if a sighting occurs.) Irregular tessalations are also utilized in, for example, spatial databases dealing with terrain data. The use of any tessalation approach requires spatial generalization to take place since each tile is normally far larger than the space occupied by a single entity and larger entities often cross tile boundaries.

In direct boundary representation coordinate data pertaining to the spatial entity are directly recorded as part of the database; for instance, the boundary coordinates of an area covered by a specific forest type. These approaches avoid the rigid spatial framework and enforced generalization of the tessalation schemes, but do so at the price of storing and manipulating much larger volumes of coordinate data. A complete review of spatial data models may be found in Peuquet (1984).

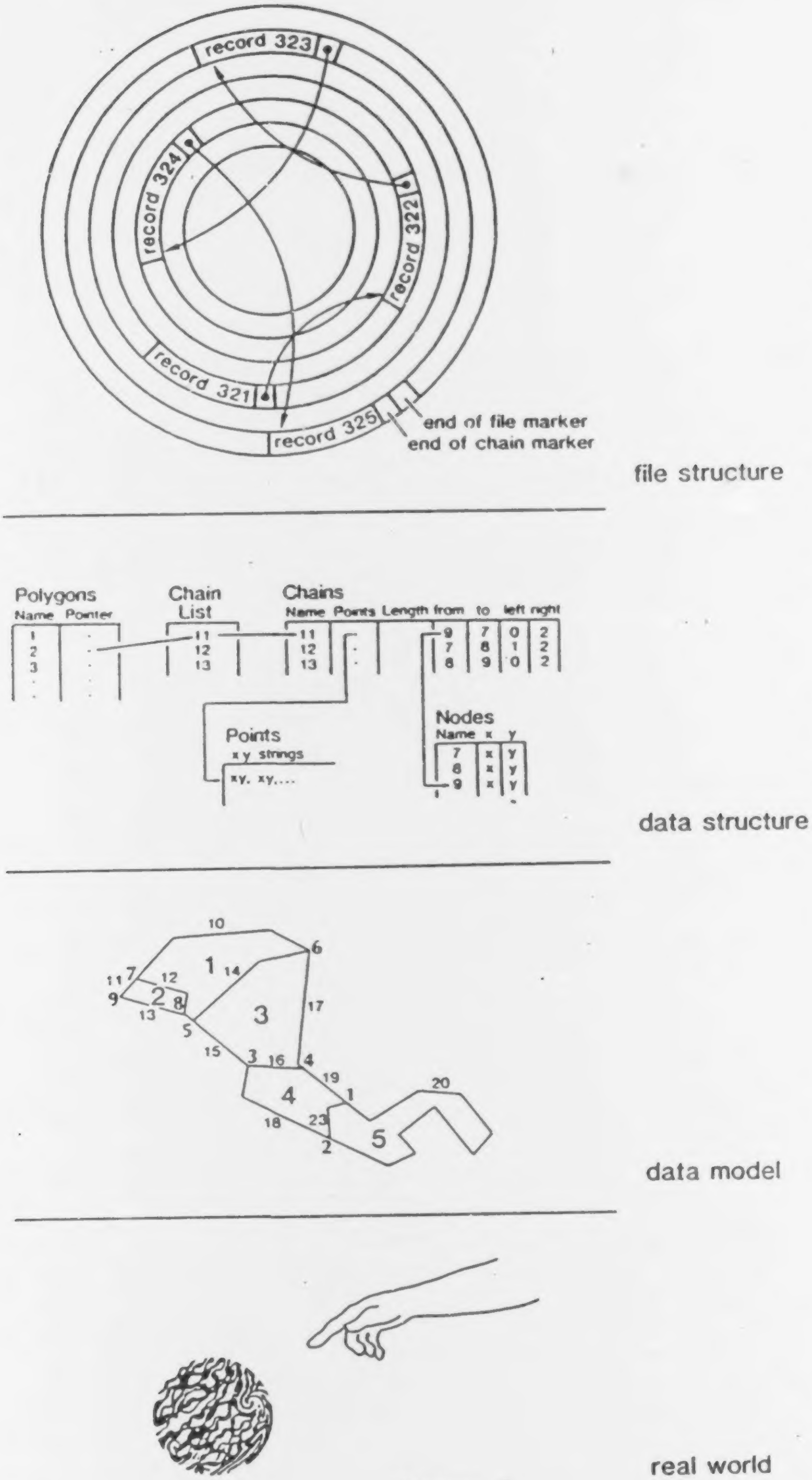
record 323
record 324
record 322
record 321
record 325
end of file marker
end of chain marker
file structure
Polygons
Name Pointer
Chain List
Chains
Name Points Length from to left right
Points
xy strings
xy, xy,...
Nodes
Name x y
data structure
data model
real world

Figure 3

Spatial Data Models and Data Structures
[from Peuquet, 1984]

## Spatial Data Volumes

The volume of spatial data increases in a non-linear fashion as the resolution increases. A very early grid cell system was developed by the State of New York (LUNR - Land Use and Natural Resource System) which subdivided the state into over 180,000 cells with nearly 200 data elements defined for each cell. The system utilized cells one kilometer on a side (subsequently determined to be much too coarse to be of more than minor utility to most potential users - a example of improper spatial generalization). Had 500 meter cells been utilized, the system would have contained over 700,000 entities and if 250 meter cells were defined there would have been nearly three million entities in the database. Unpublished estimates by the Defense Mapping Agency for a world database at the ten meter level are on the order of $10^{19}$ bytes, far larger than any database known at this time.

These very large data volumes have often restricted the application of the GIS to limited spatial areas. Many GIS applications in the United States cover only project-oriented areas (e.g., coal development areas in SE Montana) although there are a few state-level installations (e.g., Minnesota and Maryland). The largest civil system operational today, the Canada Geographic Information System (CGIS) operated by Environment Canada (see Tomlinson, Calkins and Marble (1976)), covers the developed portion of Canada and is still acquiring data after nearly two decades.

## Scale and Spatial Generalization

Every spatial database, including analog map documents, exists at a scale much smaller than the segment of the world it is designed to represent. The contents of the database must reflect the results of a selective spatial generalization of the real-world data. A major problem in spatial databases today arises out of the desire to represent different levels of spatial generalization within the same database or to change the level of generalization once the database has been created. This is possible only over a very limited range of generalizations (scales) and only in the direction of greater rather than lesser generalization. Where multiple levels of spatial generalization are required, the current solution is to create multiple copies of the database, one for each different scale. Procedures for the automatic generalization of spatial data represent a difficult and critical research frontier in the discipline.

## Problems of Designing Databases for Natural Systems

We can no more construct a database for a particular natural system by just simply starting than we can construct a complex machine without prior thought and planning. More information systems (both spatial and non-spatial) fail from lack of planning or poor planning than any other reason (Lucas, 1975 and Johnson, 1981). Failure, as used here, implies more than complete lack of functionality, it also implies partial lack of functionality which may severely impede the utilization of the data system.

Such a failure in the case of a geographic information system intended to support efforts to maintain biological diversity could lead to losses of genetic resources which could never be recovered. Proper design of the database will not insure that this will not happen, but it will insure that the probabilities are in our favor.

Over the years, a number of planning methodologies have evolved (based in part upon the experience gained in the software engineering of large programming systems) for the design of database systems. See, for example, Gane and Sarson (1979), King (1984), and Vick and Ramamoorthy (1984). Many of the concepts contained in these methodologies have proven applicable to the design of space-time databases, but additional factors must be taken into account. This section provides an overview of the activities considered basic to planning of the GIS and its associated spatial databases.

## Developing Common User Definitions: The Data Dictionary

Every database system is composed of an organized structure of entities, each of which is characterized by a number of attributes. The data dictionary is a document which spells out in technical detail how entities are defined, what each attribute is, how it is measured, who is responsible for collection and entry of the data element, etc. Without a viable data dictionary in place, the contents of the database become subject to the whims of each user who chooses to add a new element or modify an existing one. The data dictionary is the major quality control document for the database.

Often database developers and users will claim the existance of a data dictionary when it does not exist, exists only in a limited form, or is not established within an administrative structure which can insure its integrity. Such a situation can be seen in a number of existing biological databases where these conditions prevail.

In the creation of a spatial database the definition of location of the entities must be a critical part of the data dictionary. Locational factors must be spelled out in some detail since there are many levels of locational precision and many different coordinate systems in use. For example, in the United States we commonly make use of two major coordinate systems (latitude-longitude and the Public Land Survey System) which cannot be accurately translated into each other at the present time (National Research Council, 1982). One private firm has spent over a quarter of a million dollars constructing an approximate translation scheme and the NRC estimates that the construction of a high accuracy translation scheme for the United States would cost hundreds of millions of dollars.

## Establishing Internally Consistent User Views

Each data user maintains what is called a "view" or local model of the data which represents his or her notion of the data organization needed to undertake the specific work at hand. Without loss of generality, a local view may be thought of as defining the structure of a miniature database oriented only to the needs of a single user. Theoretically, there should be a local data dictionary and a local conceptual model establishing a consistent set of entities and attributes and setting forth the structural relationships between the entities. In reality, many data users do not operate on this formal level, and their view of the data that they use is a fuzzy one. This fuzzyness cannot be tolerated in a database system serving multiple users; its elimination is one of the major problems encountered in the design of spatial databases.

During the construction of a master database which is to serve a community of users, a major initial step is to work with each user to establish a local data dictionary and an explicit statement of the (often implicit) data model or models currently being used. Such investigations often reveal previously undetected inconsistencies in overall operations (e.g., data collection procedures) which must be resolved before the overall design can proceed further. Detection and correction of these local problems represents a significant, and indirect, benefit of the database design work since it can result in substantial improvements in the quality of local operations even if database creation never takes place.

## Estimation of Data Volumes

In conjunction with the development of internally consistent local views, it is critical to establish estimates of total data volumes (both present and projected) with as much precision as possible. This normally requires an in-depth examination of the current data flows including sources of data, data delivery methods, etc. Attention should also be explicitly given to any user needs for archival storage of data since this is an often neglected aspect of database design.

## Establishment of Data Manipulation Priorities

Data exists to be used and the types of data manipulation and analysis to be undertaken by the user must be established (these activities are, of course, dependent upon the needs of the user for information processing. It is not unusual for these activities to result in additional data elements being created and subsequently added to the local database. (For example, the results of a complex modelling operation may predict future changes in the spatial distribution of the phenomena being studied and the user may wish to store these predictions for comparison against, say, current spatial distributions.)

Special attention should also be given to the time distribution of data manipulation activities and to the number of times each activity is to be carried out during a typical time period. This information, as are the data volume estimates, is critical at the implementation design level of database creation since from it are derived estimates of both the scale and the temporal distribution of the stress which is to be placed upon the database system.

## Integration of User Views Into a Single Global View

The final step in the construction of the conceptual model of the master database is the integration of the individual users views into a single, aggregate global view of the database. This global view (sometimes called a "schema" with the local views referred to as "subschemas") must, of course, be internally consistent as defined by both the master data dictionary and the relationships between the entities making up the database. To achieve this unified view, individual user views must be aggregated and generalized. Frequently the process of view integration must be halted until inconsistencies in local data dictionaries and data models can be resolved.

When completed, the global view contains as logically consistent subsets all the individual, local views. It also provides the basic framework into which all future changes and additions to the database must fit. The conceptual model of the database which is created as a result of the activities outlined above represents the major input to the creation of the implementation model of the database. The implementation model is a detailed restatement of the conceptual model in more precise terms (such as may be attained through the use of a Database Definition Language (DDL)) which are closely related to the operational database systems available. The final step in the design process involves the physical mapping of the implementation model into a specific hardware/operating system environment (Teorey and Fry, 1982).

## Results of the Design Process

The database design process which has been briefly described here is based upon some simple but highly critical assumptions. The most important of these is that an overall set of goals exists for the database system and that each of the individual operations reviewed in the design process makes a significant contribution to the attainment of these goals. If an overall goal structure does not exist, then it becomes nearly impossible to resolve the multitude of differences encountered when the attempt is made to merge many local views into a unified global view.

Another critical assumption revolves around the notion that current data utilization activities, when combined with intelligent user extrapolations, hold the key to future database operations. This, of course, is closely related to the need for a clearly stated, overall goal structure noted earlier.

If these assumptions hold, and the design process is competently carried to completion, the probability is quite high that the resultant database will be both robust and successful if it is properly administered. Organizations will frequently expend substantial time and resources in the design and creation of complex database systems (both spatial and non-spatial) only to have them fall far short of their goals due to a failure to adjust the organization's administrative structure to a configuration that provides proper support. (E.g., failure to properly implement the database administrator function.)

## Approaches to Data Acquisition for Natural Systems

A major and very critical assumption in the database design process is that the data required by the database system will be forthcoming in a timely fashion and that the data streams can be subjected to whatever quality control measures are held to be necessary (these should be specified within the data dictionary). This assumption is frequently difficult to maintain when data collection operations are institutionally separated from data utilization activities.

In the area of in situ biological systems, substantial portions of the data acquisition activity are characterized by both high cost and low productivity since they have traditionally been based upon manual field operations. This, coupled with the very substantial size of the United States, has resulted in much lower levels of data availability than has been held to be desirable. In certain areas, the introduction of specialized instrumentation holds some promise for future improvements in both productivity and cost reduction.

## Traditional Approaches to Data Collection

Traditional approaches to the collection of data on natural systems have relied for many decades upon field observation by human investigators. The volume of data that can be collected in this fashion is severely limited and this has forced an operational focus upon small area studies and upon the development of a substantial statistical methodology oriented toward optimal sampling methods (e.g., Green (1979)). Even in substantial studies with a strong field orientation (e.g., Short 1983) the amount of detailed information on the spatial distribution of *in situ* biological entities is quite limited and often is not even presented in map format.

Data on those attributes of biological entities which may be examined in the laboratory instead of *in situ* are, of course, much more abundant. The most severe *in situ* data collection problem involves the initial acquisition of spatial data simply because this activity must be carried out in the field. Secondary acquisition of this data (e.g, after it has been stored on maps) is less expensive but does not remove the need for the initial field observations. (Even reduction of analog map data to digital form is a complex and often expensive process, Marble, *et al* (1984)).

A common approach to the reduction of this data collection problem is to attempt to establish close correlations between the spatial distribution of the entities in question and the spatial distribution of other factors (e.g., vegetation or forest types) whose location is often more easily determined (perhaps through the use of special instrumentation such as air photography or other forms of remote sensing). When carried out on a multi-variate basis, this approach has had some significant successes, especially in conjunction with the use of remote sensing devices.

### Instrument-based Approaches

Given the general low productivity (in terms of data generation) associated with manual field observations, it is natural to seek productivity improvements through the use of instrumentation. Instrument-based approaches have been used for a number of years in *in situ* studies, and are generally based upon the use of remote sensing devices (air photography, radar and infrared imaging devices, etc.) or radio telemetry devices where a radio transmitter is attached to the animal and radio direction finding techniques are utilized to obtain space-time data relating to the animal.

### Remote Sensing

The term "remote sensing" has been in general use for nearly a quarter of a century to signify a class of instruments which are able to collect data without being in direct contact with the subject. A number of these devices produce images which are photograph-like, but which are based upon measurements made in non-visual portions of the electromagnetic spectrum (EMS) (e.g., radar). Air photography is, of course, a form of remote sensing but the term remote sensing has been used, upon occasion, to indicate instruments operating outside the photographic portion of the EMS. A major advantage of remote sensing approaches is their ability to collect data over wide areas at a single point in time.

The single most valuable reference to remote sensing operations is the second edition of the *Manual of Remote Sensing* (Colwell, et al, 1983). Volume II contains a number of application-oriented chapters with those dealing with agriculture (Myers, 1983), forest resources (Heller and Ulliman, 1983), and rangeland applications (Carneggie, Schrumpf and Mouat, 1983) being especially relevant to the present discussion.

As might be anticipated, individual animals detectable in remote sensing imagery are generally large (e.g., seals, caribou, pelicans, etc.) even when large-scale air photography is utilized. Optimum scales appear to range from 1:8000 for moose and elk to 1:3000 or better for sheep and deer. The use of stereo photography has a significant positive impact upon accuracy of counts. There are a number of factors which significantly reduce the utility of this approach including interactions between the color of the animal and the color of the background (this may vary with the season) and obscuring effects of vegetative cover (e.g., forested areas). The use of thermal infrared scanning (which utilizes the temperature difference between the animal and its background) has apparently met with only limited success (Carneggie, et al, 1983). Radar systems have had only very limited use, but have proven somewhat successful in tracking insect swarms, etc.

Considerably greater attention has been given to remote sensing as a tool in the study of vegetation. This has largely been a result of the joint interests of the Department of Agriculture and the National Aeronautics and Space Administration in the forecasting of commercial crop yields and in supporting commercial forestry operations (Meyers, 1983; and Heller and Ulliman, 1983). Work here has focused upon the identification (often from Landsat imagery) of regions of commercial crop production or specific forest types wherein a single species (or stable mix of species) covers a significant ground area. The utility of these approaches has been high in the temperate zones of the planet (e.g., most of the United States), but they have met with considerably less success in the tropics. As Meyers (1983, p. 2263) notes: "It has been suggested that the tropical rainforest is the botanist´s dream and the forester´s nightmare. The enormous number of species and genera hampers resource assessment, initial utilization and management". This lack of homogeneous vegetation regions severely impacts the utility of remote sensing assessments.

A common approach in studying animal communities is to derive associations between the spatial distribution of the animal in question and the structure of its habitat. The habitat is often expressed in terms of some mix of vegetation types which may be easier to detect in remote sensor imagery. These surrogate or indirect measures depend upon "ground truth" investigations to establish the initial relationships.

### Radio Telemetry Systems

These systems have been used mainly for determining the behavioral patterns associated with the movement of animals within their habitat, and not for animal census operations. The instrumentation consists of a miniaturized radio transmitter (sometimes associated with physiological monitoring devices) which is either implanted in or attached to the animal. Standard radio tracking techniques are utilized to produce a record of the space-time behavior of the animal. Radio telemetry techniques have been used successfully with a number of birds and mammals (e.g., golden eagle, great horned owl, black bear, elk, etc.) and their use is growing. While providing substantial amounts of spatial detail which is unobtainable by conventional methods, their costs remain high.

### An Assessment of Current Data Acquisition Capabilities

Traditional ground-based data collection procedures are time consuming, tedious, and generally exhibit low data generation levels. Instrument-based approaches have demonstrated an ability to increase both data productivity and data quality, but only in a limited number of circumstances. Comparative costs for data collection are difficult to obtain. This is especially true with respect to remote sensing since many of the data collection activities in this area have been funded under NASA experimental programs designed to provide proof of concept. Perhaps one of the most convincing arguments with regard to instrumental approaches lies not with increased productivity and/or lower costs vis a vis tradational methods, but in their ability to provide data that simply could not be collected by traditional means.

The availability of timely, reliable data, in sufficient quantity, is very likely to be the major obstacle encountered in the creation and utilization of space-time databases for natural systems. Research on data collection methodologies, including cost and time effectiveness comparisons, should be strongly encouraged.

With respect to data systems designed to support the continuing biological diversity of plants and animals, data acquisition problems are significantly increased since substantial resources must be focused upon species composed of a very limited number of entities. These are, by their very nature, the most difficult (and costly) to identify and measure in the field since significant resources must be expended upon the inital search for the entity.

## The Current Status of Databases for Natural Systems

At the present time, a number of organizations in both the governmental and private sectors are devoting substantial efforts to the creation of databases relating to natural, biological systems. Within the Federal establishment, the work undertaken by the Bureau of Land Management and the Fish and Wildlife Service is especially critical. In the private sector, work is being undertaken by private forestry operators and especially significant contributions may be seen in the activities of some non-profit conservation groups such as the Nature Conservancy and Ducks, Unlimited.

The widespread interest in spatial databases pertaining to in situ biological systems is creating a major problem. Databases pertaining to natural systems are highly fragmented at present, and there is no central indexing scheme which is able to provide potential users with a guide to the types of in situ data which are available. For example, several hundred data sets have been created by various Federal agencies but they are held in a wide variety of locations and would be most difficult for any user to locate; they also suffer from lack of any overall data dictionary or integrated planning.

A number of years ago a similar situation existed with respect to water data in the United States; the problem of locating water-related data sets created by diverse organizations was solved by the establishment of the Office of Water Data Coordination and the NAWDEX data system which is operated by the Water Resources Division of the U. S. Geological Survey (Department of the Interior, 1980). A similar system has been established by the Canadian government to provide geographical and other indexing of oceanographic databases (Bolduc and Jones, 1983).

Recently the Federal Interagency Digital Cartography Coordinating Committee (FIDCCC) began work (through the National Cartographic Information Center of the U. S. Geological Survey) on a similar indexing system which would cover all forms of spatial data, including biological data. A different approach, with a substantial biological emphasis, is being explored by the Denver Subcommittee of the FIDCCC. (Asherin, personal communication, 1985.) These efforts, if successful, could significantly improve access to existing biological databases and, if coupled with data dictionary standardization operations, could lead to improved levels of data quality as well.

Existing databases dealing with in situ biological data have several characteristics (in addition to the fragmentation mentioned above) which reduce both their short-run and long-run utility with respect to maintaining the continued biological diversity of plants and animals:

1) Nearly all existing databases have been created in response to the demands of specific projects which are normally concerned only with small areas within the United States. (Notable exceptions may be found in the national-level databases maintained by some of the non-profit conservation groups.) The criteria for selection of these project sites has been driven by, in many cases, an interest in the economic exploitation of mineral, forest or water resources and the resulting environmental impact of these activities. Thus, there is no guarantee that the data contained in these databases will serve other goals in an effective fashion.

2) The major design problems discussed earlier in this paper are not ignored by the organizations generating the databases, but there is little standardization (for example in data dictionary terms) even within a given organization. The problem is recognized by many of the persons involved in database creation, but the combination of narrowly defined mandates for most line organizations together with the heavy project emphasis noted above makes the imposition of any broader view very difficult. Lacking any common design, it will prove to be most difficult to combine existing databases into any larger data system.

3) Each of the existing databases has been brought into existence within the context of a specific GIS. Given the

technical structure of the geographic information systems available today, it would be difficult to utilize a database created for GIS "A" on GIS "B". This represents a problem not only for the current transfer of databases between different GIS´s, but also for the archival storage of the data since the data archive may prove to be unusable if the particular GIS for which it was designed is no longer available. This archiving problem has been explicitly addressed by the Canadian government, but has received little attention from groups such as the National Archives in the United States.

It is clear that, from the standpoint of preservation of the continuing biological diversity of plants and animals, most existing spatial databases leave much to be desired from the standpoint of either short-run studies or long-range planning.

## Conclusions and Recommendations

Based upon this examination of the current situation, several general conclusions may be drawn:

1) Databases designed for examination of in situ natural systems require the utilization of special technologies due to their explicit spatial components.

2) These special technologies, known as geographic information systems, current exist and are being applied to a variety of natural systems, mostly at a small scale rather than on a national basis. Current evaluations of GIS technology indicate that research and development activities may well be required if very much larger data volumes are to be handled.

The discussion of GIS research needs contained in Dangermond, et al, (1984) discusses both the theoretical and practical needs in this area in greater detail. A major problem is the lack of focus for GIS research within the Federal establishment. As suggested in Dangermond, et al, research needs in this area and the development of a scientific plan for their accomplishment should be the focus of a National Research Council study.

3) The acquisition of the necessary spatial data is a slow and costly process despite the utilization of instrument-based approaches in some limited areas. This situation is expected to continue for some time, and extensive and immediate investigation of new data collection methodologies is urgently needed. These investigations should develop the explicit cost information which is generally lacking today since without this information the cost of assembling the necessary databases cannot be estimated.

4) Proper design of the spatial database is a necessary prerequisite to its successful implementation. Insufficient attention is currently being given to these database design problems and, as a result, the future utility of existing databases is being severely diminished. This situation is especially critical in Federal agencies since much of the current data acquisition and database construction activities are taking place in this arena.

5) Extensive fragmentation of databases currently exists and only recently has action been taken to begin development of a

partial indexing system which would provide centralized information on database composition and availability.

Examining the current situation, it would appear that certain activities could be undertaken which would significantly improve matters in the short run. Implementation of the following recommendations may prove difficult because of the large number of governmental and private interests involved, but efforts must be made in these directions if any immediate improvement is to take place.

1) The current approaches to database indexing by the FIDCCC and by its Denver Subcommittee should be carefully examined to determine if any changes should be made to better serve the interests of maintaining overall biological diversity. This activity should take place as soon as possible so that any necessary changes can be incorporated into the system at an early stage before substantial investment has been made in other structures.

2) The activities noted in (1) above are directed only toward existing digital databases. A substantial amount of in situ data on natural systems exists in analog form (maps, tables in reports and articles, etc.) which are very difficult to locate. A combined bibliographic and GIS system which permitted the spatial indexing of this material could be very valuable. (A model for this system may be seen in the spatial indexing of geologic studies carried out by the U. S. Geological Survey.) It is recommended that, if possible, the activities noted in (1) above be expanded to include this area.

3) Insufficient attention is currently being given to spatial database design. It is recommended that steps be taken to bring the principles of proper design to the attention of organizations (both governmental and private) engaged in the creation of these databases. The development of data dictionaries which are acceptable to a broader range of organizations would increase the future utility of these databases and could be brought about through joint government/private studies coordinated by the National Research Council.

4) Data acquisition is expected to pose continuing problems. Organizations concerned with research on natural systems (e.g. the National Science Foundation) should be encouraged to support research and development activities leading toward increased productivity in this important area. Similar encouragement must also be given to focusing and supporting research and development work on geographic information systems.

## Notes

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# Appendix A

## Biological GIS Applications at the Bureau of Land Management[1]

### WILDLIFE HABITAT MANAGEMENT

1. Modeling Potential Pronghorn Antelope Range

   The GIS has been used to model the interaction between land cover classification (vegetation inventory) and administrative habitat management units. The resulting classification of high, medium, and low areas of concern for managing pronghorn antelope range was developed by using vegetation stratification and acre tabulations. Map products were prepared for resource management plan conflict analysis.

2. Potential and Critical Mule Deer Habitat Modeling

   The GIS has been used to model the interaction of grazing allotment boundaries, vegetation inventories, and terrain (slope) data to determine mule deer management opportunities and constraints. This effort provided valuable data used for resource conflict resolution and decision making, which consisted of acreage tabulations and map products at various scales.

3. Edge Effect Modeling for Maximizing Wildlife Use Zones

   Using surface cover data as input, the linear contact between specific vegetation types was calculated--a zone commonly referred to as the edge effect. The potential increase or decrease in edge effect from proposed surface disturbances (mining, logging, fuel wood cutting, building range improvements) has been used in habitat management proposals and mitigation discussions in activity plans.

4. Raptor Habitat Buffering

   Using known locations of raptor nesting sites and slope, aspect and other terrain data, raptor habitat was modeled. The resulting data layer has been used in proposed mining and logging mitigation plans and in estimating production losses (acres of timber, tons of coal).

5. Aquatic Habitat Modeling

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[1] Compiled by Mr. Robert Ader of the Division of Advanced Data Technology, Denver Service Center, Bureau of Land Management. The list is most likely not complete since several BLM field offices have had GIS capabilities for several years and applications have not been completely tracked.

Riparian zones were modeled using stream network data and buffering capabilities to produce linear measurements and analyze conflicts for managing aquatic habitat. Maps at various scales and calculations of acres and miles were produced.

6. Impact Analysis of Oil and Gas Exploration on Wildlife Habitats (elk, deer, grizzly bear, desert bighorn sheep, prairie dog towns), Migration Routes, Bedding Grounds, Calving Grounds, and Winter Range

The impact analysis process is important to the resource specialist. The GIS´s capability of testing the "what if" questions has increased the interest in interacting with other related databases for conflict resolution. The GIS has provided a means to quickly recommend mitigation to a resource manager. Power lines, gas lines, and road construction have been tested through the impact analysis process using GIS at a number of locations.

## RANGE MANAGEMENT

1. Vegetation and Land Cover Classification

Early remote sensing efforts to create land cover classification data layers and maps were mainly designed to assist in stratifying the vegetation component of renewable resources. Stratification permitted field survey crews to direct their energies to areas of demonstrated importance, making field inventories more efficient. These data were received with limited success, but the remote sensing effort did show that graphics data could be digitally manipulated to provide field offices with a computer-assisted tool. The BLM is using the results of field mapping and digital analysis for allotment management plans, activity plans, environmental assessments, and resource management plans. Of particular benefit to inventories is the creation of tabular summaries and hardcopy maps. The GIS has only recently been used for the subsequent analyses.

2. Expected Grazing Pressure Model

Vegetation areas receiving heavy to no grazing pressure are traditionally mapped as range suitability zones for livestock management. Generally, range suitability considers watering points, distance from water weighted by terrain (slopes) influences, watershed condition, and the vegetation component. The expected grazing pressure model incorporates only the physical criteria of distance to water and slope limitations to model zones of livestock influence. The resulting data layer may then be used in a variety of analyses.

3. Range Vegetation Stratification for Treatment

To increase forage production, field offices must control unwanted vegetation. Such control, however, is limited by the terrain. By modelling the interaction of guidelines for the use of control procedures, the BLM can produce information that can be used to select mitigation measures to protect other resources. The cost estimates, acreage tabulations, and hardcopy maps are a few of the products.

4. Range Improvement Siting

Water development and allotment boundary data layers have been overlayed to assist in mapping pipelines, trough locations, wells, springs, stock tanks, and other range improvements. After these data layers are overlaid with livestock service areas, unused forage outside these areas may be plotted and mapped for potential range improvement activities. Adding new fences, salting grounds, or waters may be evaluated by engaging GIS generated alternatives for maximizing forage availability.

5. Impact Analysis from Surface Disturbances

Impact analysis is a standard process for evaluating livestock forage losses. Impact analyses for the proposed actions for oil and gas wells, mining, roads and pipeline construction require maps and acreage tabulations to assist in making resource management decisions.

## TIMBER MANAGEMENT

1. Fuel Wood Cutting Area Modelling

The GIS is being used to model the relationship of the vegetation inventory databases and terrain (slope) data to provide a stratification for potential fuel wood cutting areas. The demand for fuel wood in the pinyon-juniper woodlands requires an inventory of potential cutting areas whose slopes allow the use of light trucks. To assist field timber specialists in locating potential areas, various databases (land ownership, road networks) are used to create maps and tabular summaries to begin the field inventory.

2. Timber Harvest Suitability

The relationship between terrain (slope) data, timber compartment, and perennial stream data has been used to create maps and tabular summaries for timber harvesting. Slope limitations are modeled to determine the best means (tractor, cable, or restricted) to harvest timber that is more than 100 feet from perennial waters. Tabular summaries and maps of timber compartments, stands, and types are crated for each harvest suitability class.

## SOILS MANAGEMENT

1. Soil Inventory Database Edit and Map Creation

The Soil Landscape Analysis Procedure recently tested by the BLM, Geological Survey (EROS Data Center), and the Soil Conservation Service, has used GIS to assist in soil mapping and to aid field crews in creating accurate and consistent databases. The procedure uses editing functions for maintaining continuous soils data. The develope the GIS soil database, the procedure uses the field mapper's expertise and the terrain (slope and aspect) data to assist in the pre-mapping function of field soil survey procedures. The soils data layer is then put together with other information to create soil correlated and derived data layer. The resulting maps and tabular summaries are then used by the soil mapping field crew.

2. Soils Inventory Data Layer

The base maps for soils are typically the first data layers to be digitally entered in to the GIS system because of the land manager's need to correlate recommendations to the soils component. Generally, if the soils data layer exists, most disciplines will use it to create a supplemental thematic data layer.

## FIRE MANAGEMENT

1. Fire Fuels Mapping

The fire management program has recognized the need to create a fire fuel data layer. Fire fuels are mapped using vegetation, slope and transportation networks. The initial attack for fire suppression uses the road network and fire fuels data layer to direct crews to the field.

## WILDERNESS STUDY AREAS

1. Wilderness Study Area Inventory

The boundary proposals of wilderness study areas have been captured and then overlain with resource data layers to identify potential conflicts and intrusions. The output maps and tabular summaries have been used in wilderness study reports.

2. Legal Description for Wilderness Boundary

The GIS is being used to provide legal descriptions of wilderness study area boundaries for publication in the Federal Register. The interaction between two computer systems is providing field offices with required distance and bearing legal descriptions.

## WILD HORSE MANAGEMENT

1. Wild Horse Management Plans

The GIS has been used to help prepare wild horse herd management plans. The wild horse range boundary is entered into the database and overlain with the other data layers to create maps and tabular summaries for wild horse management plans, and to produce output for resource management plans.

Contract Paper No. 3

# Planning and Management Techniques for Natural Systems

Herman H. Shugart
University of Virginia

October 1985

Office of Technology Assessment
U.S. Congress

This contract paper was prepared for the assessment of "Technologies to Maintain Biological Diversity." The views expressed are those of the author(s) and not necessarily those of OTA.

OCT 7 1985

# Planning and Management Techniques for Natural Systems.

A report to the Congress of the United States, Office of Technology Assessment for the assessment of Technologies to Maintain Biological Diversity.

by:

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# ABSTRACT

One element of preserving the species diversity of large areas is to manage the habitats of organisms in the areas to insure the survival of all of the indigenous species. Two methods of conducting such management are to either manage the landscape for a diverse array of habitats (area-diversity management) or to manage for particular species (featured-species management). Central to either management approach (and to other approaches that represent mixtures of the two management strategies) is the knowledge of the nature of the association between organisms and habitats. Traditionally, information has been collected at two diferent spatial scales. Most of the applications of habitat management strategies intended to insure a maintenance of species diversity have not considered the dynamics of the habitat as it changes over time due to natural (or anthropogenic) disturbances. There exist a number of computer models of vegetative dynamics and these models could be used as prototypes to develop habitat simulators to be were used in diversity management. Extant cases of such applications appear to be relatively successful. There is a need for field testing of the methods of habitat management that have been developed. The tests that have been conducted to date indicate that the methods work well in cases in which there has been a sufficient amount of information collected to form the management model.

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# INTRODUCTION

Species diversity exists at different levels of spatial organization. The late R.H. Whittaker (eg. Whittaker 1975) recognized and named three levels of diversity. First, he identified alpha-diversity which he defined to be the diversity of species on a particular tract of land. Second was beta-diversity, the number of species in a particular landscape or large area. Third was gamma-diversity, the richness of species over a large region. These three types of diversity can arise from different underlying causes. For example, the alpha-diversity of a small area might be largely a consequence of the habitat in the area. In the same example, the gamma-diversity for the region could be a consequence of the geological history and geography of the region.

The three levels of diversity are interelated and plans designed to conserve diversity must consider multiple spatial scales. Otherwise, one could modify several small areas each to have maximum alpha-diversity but find that if the species that made up this alpha-diversity were the same for all patches then the regional or beta-diversity to lower than that of a landscape composed of a mixture of patches with a less species of different kinds in each patch. It is particularly important not to focus so strongly on creating fine-scale diversity to the degree that the larger-scale diversity is lost from consideration.

Species of plants and animals are being lost at a rate that has been increased by several human-related factors. Some of these species are attractive to the general public (for example, large terrestrial vertebrates or plants with showy flowers) and for this reason gain a considerable degree of public enthusiasm toward the insurance of their survival. However, many of the species that are in danger of extinction are not attractive in the same sorts of ways (for example, beetles or mites) and the interest in their preservation is not nearly so intense. The successes in managing for rare and endangered species have been the greatest in the cases in which the knowledge of the species' life history attributes have been considerable and in which public attention has caused funds to be allocated to implement a scientifically sound species management scheme.

At the global level, the knowledge of even the most elementary information is lacking for many of the species that make up the world's diversity. Some species from tropical areas are in danger of becoming extinct before they are even described - much less understood in terms of their life histories. Minimally, we must continue to manage intensively for the rare and endangered species that are already being protected and we hopefully will expand these efforts to other species that are in danger of extinction. If we are to preserve the world's biotic species-diversity, we also will need to take action to maintain other species that are less in the public eye and that are not well known scientifically. To do this, we must develop the ability to manage landscapes to preserve entire ecosystems in the hope of increasing the likelihood of preserving the organisms that make up the biota of these systems. In this way, it may be possible to assure the continued survival of the species that contribute to the overall global diversity.

This report discusses the key elements involved in developing an ecosystem management approach to the maintainence of the diversity of species. In doing so, it will draw examples of elements of such an approach from actual cases in the United States and other nations. Primary sources for these examples will be taken from the "open" scientific literature. These sources have been subjected to rigorous review by expert and disinterested peers and are readily available in libraries in the United States and elsewhere.

## THE IMPORTANCE OF HABITAT

One of the most important factors contributing to the survival of any species is the continued presence of suitable habitat for the species over time. Habitat is a well-developed concept in modern ecology and wildlife management, and is defined as the kind of place(s) where individuals of a plant or animal species can survive, grow and reproduce. A species' habitat is frequently quantified in terms of the ranges of physical and biotic factors associated with the presence of individuals of the species. For many animals and particularly for vertebrates (Morse 1968, Shugart and Patten 1971, Dueser and Shugart 1979, Franzreb 1983) the structure of the vegetation is a major element in determining the suitability of the habitat (Hilden 1965). Because the vegetation can be altered by both natural and human-controlled processes, the ability to project vegetation change over time is an essential element both of managing species' habitat on a continuing basis and of minimizing the potential extinction of certain species.

Even though habitat is a long-established ecological concept, there has been a considerable amount of development of the habitat paradigm in the last 2 decades. This development was manifested initially as improved descriptions of habitat (in the form of precise mathematical formulae) and secondly in the form of an increased ability to predict change in the vegetation. In both cases the development has been catalyzed to a great extent by the increased availability and increased power of high-speed digital computers.

## HABITAT MANAGEMENT: CONCEPT DEVELOPMENT AND EXAMPLE CASES FROM USDA FOREST SERVICE (UNITED STATES)

In the early 1970's the USDA Forest Service became engaged with the problem of managing public lands to insure the maintenance of diversity (along with several other objectives in the overall rubic of supporting research toward a multiple-use forestry practice in the United States). The agency sponsored open symposia (e.g. Smith 1975, Capen 1981) that discussed scientific problems as well as appliations and helped to catalize a innovative period in exploring habitat management techniques. Two different management approaches developed that each were associated to a great degree with the economic and political constraints attendant to the land management problems in two different regions of the United States. These two concepts were the "featured-species plan" (Zeedyck and Hazel 1974) used in the productive forests of the southeastern United States and the "area-diversity plan" (Evans 1974) used in the hardwood forests of Missouri. In both cases, detailed instructions were developed that determined the management procedures to be used on a given area of land.

## The Featured-Species Plan.

In the featured-species plan (Zeedyck and Hazel 1974), a species of bird or other animal is chosen to be featured by the management plan. Studies are initiated to better understand the ecological requirements of this featured species and a plan for forest management developed that optimizes the availablity of habitat for the featured species and economic factors that normally would be involved in timber harvest. This mode of management suits the situation in productive forests in which the generation of habitat for the featured species (in some cases) can be included as part of the thinning and harvest cycles used to manage the forest. Once the featured-species management plan is in place efforts can then be made to also increase the habitat suitability for other species that were ecologically associated with the featured species.

The featured-species plan developed in a region of production forestry under economic constraints that mitigated against the removal of land from production. The plan involves relatively intensive directed management and a reasonable knowledge of the habitat requirements of the featured species and associated species.

## The Area-Diversity Plan.

The area-diversity concept (Evans 1974) is designed to create an optimal diversity of animals by managing land to insure the prescence over time of all of the recognizable habitat types in a region. The plan evolved in Missouri in a hardwood forest region in which all but the most productive forests were used for recreation and watershed protection. The area-diversity plan uses the treatment of blocks of land (ca. 40 ha) to maintain a landscape mosaic of different habitat types. Tracts of land are cut and even-aged forests are regenerated on the cut-over sites. The amount of land cleared in any given year is designed to initiate a sequence of tree regeneration and regrowth leading to mature forest. Such a system is very conservative in terms of how much land might be harvested in a given year. For example, if it requires 200 years to develop a mature forest (which is a short period of time for such development in many ecosystems), then the amount of land that could be cleared in a given year would be one two-hundredth of the total forest area or 0.5%. Harris et al. (1984) has designed a similar sort of plan to combine timber and wildlife management in the Douglas-fir forests of the Pacific Northwest.

The reasoning behind these plans is that only by insuring the sustained presence of all habitats over time can one hope to maintain the diversity of the species associated with a given ecosystem. The area-diversity plan uses ecological succession as a central principle. The plan developed in a region in which there was less commercial timber production and in which there was considerable interest in maintaining the biotic diversity (which due to the recreational nature of the forest was a valuable attribute). The plan requires a considerable knowledge of the vegetation dynamics and an understanding of the habitat requirements of a large number of species.

## Synthesis between Area-Diversity and Featured-Species Plans.

The management plans discussed above as well as others (Gill et al. 1974, Holbrook 1974) have evolved considerably since the early 1970's. For example, a management plan that was essentially an area-diversity plan might use a featured-species plan on areas that harbored a rare and endangered species or an important game species (Zeedyck and Evans 1975, Harris et al. 1984). Nonetheless, the plans do represent extremes in a spectrum of land management plans that have actually been applied in practical situations. Key elements of both of the plans include the association of plant or animal species of interest with habitat and the understanding of the dynamics of habitat (and the consequent need for an active and sustained management plan). These elements can be augmented technologically by the use of computer models to actually simulate the dynamics of the vegetation and by the use of more accurate descriptions of habitat resulting from the application of modern statistical techniques. The most important aspect of these habitat management plans is that they form a background in experience and concept for a landscape/ecosystem management scheme that can be used to manage large areas to conserve diversity.

# ECOSYSTEM/LANDSCAPE MANAGEMENT

The featured-species and area-diversity concepts have some of their origins in the mission of the developing agencies, which in the case of the USDA Forest Service is strongly oriented toward forest utlization for timber. An important difference is one of scale: The featured species concept is essentially a fine-scale technique that allows one to tailor a tract of land for the apparrent needs of particular species; The area-diversity concept is a larger-scale technique that is more oriented to maintaining regional-scale diversity. One can see elements of an ecosystem/landscape management plan in such plans. This section will identify the key elements of an ecosystem/landscape management plan for regional species diversity and provide examples of the actual application of these elements. The key elements of are:

1. The ability to associate individual species with the features that make up a landscape. In the absence of detailed information on individual species, the development of theories as to what should be expected in terms of patterns of organism/environment associations is needed.

2. The ability to predict the changes in the character of the landscape. This understanding of the dynamics of landscapes ideally should allow the potential of forecasting the response of the landscape to both natural and novel disturbances.

3. An understanding of the species will respond to a dynamically changing landscape. In the absence of experimental studies to provide insight into the species response to a

changing system, there is need for a theoretical basis that would allow one to proceed in developing a management plan. This understanding should be both species-centered (how the dynamics of the ecosystem might effect a particular species of interest) and ecosystem-centered (how the success of failure of a given species might effect other components of the ecosystem).

Each of these key features will be discussed and examples of management applications featuring aspects of these key features will be provided below.

## Key Element 1: Associating Species with Landscape Features

Models that relate organisms to habitats or other features of the landscapes form a diverse array of methodological approaches and a correspondingly large scientific literature. The more recent cases are dominated by statistical approaches. These cases differ most clearly with respect to the scale in space to which the methods can be expected to apply. A dichotomy of large- and small-scale methods will be used to frame a general discussion.

### Large-Scale Approaches

For animals, there is a large repository of information that associates the abundance (or potential abundance) of a species with the condition of a large tract of land. In general, one uses a high density of a given species as strong evidence for the suitability of a given sort of habitat. This may not always be the case. For example, Van Horne (1983) proposed that habitat quality be defined in terms of fitness (an index that increases with the probability that a given individual organism will produce viable offspring) and noted several cases in which high densities in a given habitat did not necessarily indicate a high fitness. One such case occurs when large numbers of subdominant individuals are forced into marginal habitats by more dominant individuals during episodes of high population densities. In such instances the densities in what is essentially marginal habitat can be misleadingly high. Such cases have been documented for small mammals (Kock et al. 1969, States 1976, Schantz 1981) and birds (Fretwell 1969, O'Connor 1981). These considerations speak to a need for more insight into dynamics of the populations as an aid to better management.

In general, any methodologies that use presence or abundance of a species to determine habitat quality and then use such information in habitat management are based on correlations. Thus, because an species occurs in a habitat called "hardwood forest," provides a reasonable expectation that it might occur in a managed hardwood forest, but this expectation is based on the strength of correlations between the apparent animal abundance and the features of the gross habitat over relatively large tracts of land.

The potential problems that arise from using correlations to develop management plans can be illustrated by the following example. Through most of the 1970's, studies in the United States that attempted to relate the abundance of birds with habitats used vegetation sampling schemes that quantified the structure of the living vegetation. These vegetation surveys typically did not take into account the presence of old dead trees ("snags") that frequently contained holes that could be used for nest sites by obligate hole-nesting birds. Thus, it was possible to develop management schemes that created forests structurally like the seemingly prefered habitat of a species (measured in terms of the living vegetation) but that contained no nest sites for the species. The removal dead and dying trees is a standard forestry practice in almost any forest system and the acceleration of timber harvest also diminishes the abundance of snag trees. Only by an explicit recognition of the importance of snags and dead trees as a part of the habitat (e.g. Conner et al. 1976, Evans and Conner 1979, Thomas et al. 1979, Zarnowitz and Manuwal 1985) could habitat management schemes be improved (e.g. Conner 1979, Thomas et al. 1976) by leaving snag trees to provide nest sites for cavity-nesting birds.

The problems that are potentially associated with the use of correlational relationships to design management schemes to insure a maintainence of diversity sound a cautionary note and underline the need for continued studies to improve management plans. It is likely that correlations will form the basis of almost any management plan in its initial development stage. This is because correlational data is the most easily obtained and has the greatest potential to demonstrate positive results after a modest initial study (almost always an administrative prerequisite to continued support). It is important that one recognize the potential short-comings of using correlations and plan to continue research that can be used to improve any management plan developed on a basis of correlations. The minimal level of follow-up studies would be the continued monitoring of the managed system to determine if the management plan was producing the desired results.

## Small-Scale Approaches

An alternate method of associating a species with habitat is to correlate the presence or abundance of the animal with habitat features measured at a very fine spatial scale (typically 0.1 ha or less). The essential observation behind these small-scale approaches is that a species found in, say, "Oak-Hickory Forest" are not necessarily found in all parts of such a forest. The habitat of the species is characterized by the unique features of the subset of the area where individuals of the species are most frequently found.

This approach explores the correlations among the species occurrance and the quantitative measures of the habitat. The statistical methodologies used are often based on multivariate models that can take into account the intercorrelations that are a prominent feature of sets of habitat variables. Unfortunately, these methods also tend to render the results of such analyses to a form that is not easily understood by most land managers. The small-scale approaches have been the topic of a recent symposium (Capen 1981) that provides several examples (including applications) of the range of methods that are in use. Green (1980) reviews the multivariate statistical methods used in these procedures and provides 134 references documneting their used in an extremely wide range of species. While the majority of applications of these methodologies are for animals there are also cases of using the techniques to determine plant habitat (Strahler 1977 & 1978, Gehlbach and Polley 1982, Mann and Shugart 1983).

A typical hypothesis used in these studies is, "Given: (1) that Species A occurs in this area , (2) it could (in theory) occur anywhere in the area, and (3) that it (in fact) is found in only part of the area, the common features of the places where Species A is found are important to the species." There is an element of capitalizing on the heterogeneity of pattern in habitat to understand in greater detail the factors controlling the distribution of a species. Nonetheless, these studies are almost always correlational in nature and the level of detail is not a substitute for actual experimentation to determine with factors are important to the species.

## Key Element 2: Vegetation Simulators as Dynamic Habitat Models

There are several computer models designed to predict the dynamic change for a wide range of vegetation types. The implications of these models for the understanding and management of natural landscapes are great but the present discussions will focus on the potential of these models to project the change of habitat in response to vegetation change. A way of organizing the discussion of these models as habitat simulators is to separate those models that simulate the qualitative change in the vegetation on a small patch of a land over time and those that simulate the dynamics of the dominant plant populations on a small patch of land. Both sorts of simulators are logically compatible with a landscape mosaic concept of ecosystem dynamics. The expected change in a landscape over time is obtained by summing the response of a large number of patches as simulated by such models.

## Vegetation/Patch Simulators.

Vegetation/patch simulators project the change in landscapes at the scale of 100's of hectares and in some cases 100's of square kilometers. The models tend to be mathematically simple relative to the individual-plant simulators that are discussed below. Markov models of landscapes are a typical example. Markov models are based upon certain assumptions in their application at the landscape level:

1. They are based on the assumption that a landscape can be divided into a set of discrete patches.

2. Change in the vegetation is expressed as a set of probabilities that the vegetation will be in some other state at some fixed time later, given the state of the vegetation on a patch at the present time.

3. The probabilities of change are assumed to be constant over space and time.

The state of the vegetation on a small patch of land is usually characterized as to forest type or categorized by the species of the most dominant trees growing on the patch.

A typical parameter in a Markov model is a transition probability - the proportion of stands that are currently dominated by one plant species that will be dominated by another species after some time interval. It is a definite advantage of the Markov formulation that the parameters are straight-forward in terms of what they represent. One can use remeasured vegetation survey data to obtain the transition probabilities (although it requires a considerable amount of such data to estimate the parameters of a Markov model that has very many states). In several cases, the parameters of Markov models have been estimated by inference. For example, Horn (1975) assumes that the percentage of young trees growing beneath large canopy trees of a given species could be taken as an estimate of the transition probability in a Markov model. Waggoner and Stephens (1970) provide an example of using resurvey to parameterize a Markov model for a forest in Connecticut.

To implement a Markov model on a computer, one uses as a starting point the a data set that represents the initial vegetative categories of each of a large number of patches that represent the landscape of interest. A typical starting point might be the landscape as sampled from a survey. The computer would then generate random numbers that would be used in conjunction with the transition probabilities to determine the state that each of the patches would be likely to be in at the next time of interest. Because there is a element of chance as to how each patch will change over the time period of interest, there is some variation in the predictions of a Markov model from one computer run to another. Normally the models are run several times (called a "Monte Carlo" simulation) and the results are averaged to obtain an estimate of the expected behavior of a landscape and the range of variation in the prediction. Because these models have an emphasis of book-keeping the condition of many patches and because the models do not require elaborate computational steps, they can be solved rapidly on a digital computer. They generally can be implemented with ease on a typical "PC"-sized computer.

## Individual-Plant Vegetation Simulators

Individual-plant vegetation simulators in current usage today also frequently are developed with the view that the landscape can be considered to be a mosaic of patches where each patch can be thought of as having some description as to the type of vegetative cover. Unlike the example Markov models, the small-scale models are constructed to simulate with some degree of detail the ecological interactions within each patch. Frequently, the models compute the dynamics of individual plants within each simulated patch and produce predictions that are quite detailed. As an example of a small-scale vegetation simulator, one might consider gap models of forest dynamics (Shugart 1984).

Gap models function by computing the diameter increase of each of the trees growing on a small simulated plot around 0.1 hectares. The trees are increase in size annually . Mortality of the trees on the plot and the birth of trees are stochastic functions that are also applied annually. The fundamental approach in developing these models was outlined by Botkin et al. (1972). Shugart (1984) provides a detailed explanation of the model assumptions and relates the behavior of a several gap models to modern ecological concepts of forest dynamics. The models do not require elaborate data sets for parameter estimation. The philosophy that underlies the development of gap models is to represent the dynamics of the forest using general equations that can be obtained from basic physiology, morphology and silvics of trees. The models have been successfully tested against independent data in several cases and appear to do a reasonable job of predicting the pattern and dynamical change in forests.

The implementation of a gap model on a computer is much like the one described for a Markov model. Typically one would start with a set of surveys that characterized plots located on a landscape and then would use a computer to project the dynamics and change of each of the plots. The main difference in the two approaches is that while the change of vegetation type on a plot in a Markov model is determined from a single set of transition probabilities, the change in a gap model is computed by predicting the birth, growth and death of each of the trees on a plot. Output from a gap model resembles typical vegetation survey data in its level of detail. The models produce a list of the size and species of every living tree on the simulated plot. The greater complexity of the gap models does not prohibit the use of the models on PC-sized computers but the models are more often implemented on larger "main-frame" computers.

## Trade-Offs Between the Two Approaches

Both the large- and small-scale models have certain advantages and disadvantages in terms of their application to landscape management problems. On the case of the example model types above, the gap models run more slowly on a computer and require a research scientist with some training in ecological modeling in their development stage. Gap models can be used to produce a detailed description on each of the plots that sample a landscape and often have a sufficient degree of realism that they can be used to predict the response of a landscape to a novel change (e.g. climatic change, air pollution stress, change in wildfire regime). Markov models are computationally fast and fairly easily set up on most computers. They can be used to generate dynamically changing maps. Markov models require a considerable amount of data for parameter estimation, particularly in complex systems and they have no internal mechanisms in their formulation that would allow them to reasonably be expected to predict system response under novel conditions.

The most logical evaluation of the question,"Which model should be used?" is to say, "Both." The two modeling approaches are compatible and a wise approach to using the models to predict landscape change would be to develop a nested set of models with the gap models being used under new conditions and the Markov models being used for the large scale projection of landscape change. The gap model output is more compatible with the scales used in multivariate statistical studies of animal habitat selection; the Markov approach leads one to attach more general habitat descriptions of the sort used in field guides to indicate where a species might be found. The discussion to this point has used specific types of large- and small-scale models as examples of a larger set of models of each kind.

## Key Element 3: Organism and Landscape Interactions

An important ecological concept is that landscapes can be thought of mosaics composed of small distinct patches that change in their character over time. This is a venerable concept in ecology (Aubreville 1933 and 1938, Watt 1925 and 1947, Whittaker 1953) that has recently emerged as a strong unifying concept in forested ecosystems (Bormann and Likens 1979a&b, Whitmore 1982, Shugart 1984). Bormann and Likens (1979a) noted that forest ecosystems "... may be visualized as an array of irregular patches composed of vegetation of different ages. ... The structure of the ecosystem would range from openings to all degrees of stratification, with dead trees concentrated on the forest floor in areas of recent disturbance. The forest stand would be considered all-aged and would contain a representation of all species, including some early successional species, on a continuing basis." Watts (1947) classic paper documents this same concept for a wide range of ecological systems including not only forests but bogs, heathlands, grasslands and alpine communities.

The use of multivariate statistics to quantify the relation between organisms and habitats fits naturally with the mosaic concept of a landscape. This is because, in the design of the multivariate techniques, the research scientist often attempts to capitalize on the fine scale heterogeneity of the landscape to provide insight into patterns of habitat selection. For this reason, the samples used to provide the basic data for the multivariate analysis often are collected with the explicit goal of having spatial dimensions that are analogous to those of the pattern of the spatial mosaic and that thus match the scale of the elements of the landscape mosaic.

If the scale of a landscape simulation model and the scale of the data used to calibrate a species/habitat relationship are compatible, then one can meld the two to simulate the dynamics of populations in response to a dynamically changing landscape. There are potential problems that could lead to error in using such an approach that could stem from several sources. Errors estimating the nature of the association between the habitat and the species presumably can be minimized by increased sampling. Errors due to the inabilty of a landscape model to exactly reproduce the conditions on the modeled landscape is aaother primary source of error that can probably be minimized by a rigorous model-testing/model-reformulation protocol. A third important source of errors could arise from important interactions not included in the model. These errors could be due to spatial effects not included in the model (for example, the dispersion of a species across a landscape might be so slow as to prevent the colonization of newly generated habitat before it disappearred due to successional processes. Another source of the errors outside the model formulation could be due to interactions with other species that are also using the landscape (competition, predation, parasites harbored in one species infecting another species). Control of these sources of error require a concerted effort in basic ecology particularly focused on environment/organism and organism/organism interactions.

Even in the face of several sources of error and a definite need to better understand basic ecological processes, there have been some notable succeses in using the elememts of a landscape/ecosystem management concept to manage to conserve diversity. These examples will be discussed in the next section.

## EXAMPLE CASES

The example cases include three categories: 1. Static cases in which the relationships between the organisms and the habitat are used to manage for species diversity but in which change in the habitat is not a major aspect of the management plan; 2. Cases using a Markov model to simulate the habitat dynamics; 3. Cases using more detailed models to simulate the habitat change.

### Static Approaches

Because the static techniques involve using correlations between organisms' occurrance on study sites with the vegetation structure and other environmental conditions on the sites, the methods are sensitive to the amount of calibration data and to the range of conditions spanned by the calibration cases. The methods could be expected to improve with additional experience but the level of reliablity of the indices is directly related to the amount of effort spent in developing the basic data sets.

The limitations in the applications of the methods other than due to sampling intensity have not been explored to any great extent and may be masked to some extent by the need for better data sets in the case of most applications.

Case 1: HEP Models

The use of management of the habitat of organisms is central to the design of almost any land management plan that would take into consideration the response of different species to land-use change. The tremendous number of environmental impact statements that have been prepared to assess the effects of various public works on species diversity represent a repository of examples applications of a static approach to organism-habitat management. A number of techniques called HEP (Habitat Evaluation Procedures) have been developed to "automate" the impact evaluation process. The HEP techniques are essentially data management techniques that use a computer to cross-compare the effects of a given management procedure (expressed in terms of gross change in the habitat) and the general requirements of each animal species that might potentially occur in a region (Schamberger and Cordes 1985, Fry et al. 1985). These techniques in their present application use relatively general accounts of the requirements of the various species. These accounts are similar to the brief statements found in field guides to describe an animal's habitat. In some cases they are based on expert opinion.

The methods have been evaluated in some cases to obtain an idea of their reliabilty. There appears to be some variability in the formulation of the underlying model when different calibration data sets are used (Stauffer et al. 1985). The tests of the procedure involving the occurrance or absence of a species on a tract of land following a habitat alteration in general are relatively successful (Raphael et al. 1985, Dedon et al. 1985, Fry et al. 1985). Attempts to use the procedures to predict the actual relative abundance of animals have proven much less successful (e.g. of 650 model predictions of relative abundance of birds, mammal, reptiles and amphibians between different successional stages, 43% were in error, Raphael et al. 1985). The methods should evolve as more effort is directed to improving the ecological details of the individual species habitat requirements (Toth et al. 1985, Brennan et al. 1985) and as the successes and failures of the models provide insight into the parts of the models that need improvement (Raphael et al. 1985).

### Case 2: Riparian Bird Habitat Management in the Colorado River

The potential of these static approaches when sufficient detail is added to the models is demonstrated in studies conducted in the Colorado River riparian zone. In these studies, a model was developed by using multivariate statistical techniques to quantified the animal-habitat relationship. This model was able to predict relative abundance for several species with a 90% rate of success in two tests (Rice, et al. 1985). The model was developed as a part of a project in which the investigators took as a goal the design of an artificial habitat (created on dredge spoils in the Colorado River) that had a higher species diversity per 40 hectares than any naturally occuring riparian vegetation in Arizona. They successfully attained this goal and also developed a computer program that allowed the assessment of the changes in presence and abundance of about 70 species of birds in response to different management strategies on dredge-spoil vegetation. This successful program combined the use of multivariate statistics to analyze bird habitat-use with well-designed field surveys of vegetation and bird populations. The applicability of the model was regional in its scope.

### Case 3: Locating Potential Endangered Species Habitat from Satellite Imagery in Australia.

One static approach to the evaluation of habitat with a great potential applicability is the use of satellite imagery to locate potential habitat for species. One example of this application (Saxon 1983) was able to use satellite imagery to survey for potential sites for the re-introduction of the rufous hare-wallaby (Lagorchestes hirsutus), a rare Australian marsupial currently known to occur in one location in a 65,000 square kilometer area . Previous field surveys over a 10 years over a 3.75 million square kilometers area associated with a wildlife preserve had failed to reveal any potential sites. The satellite survey technique located three sites of which two were of sufficient size to consider as release sites.

## Vegetation/Patch Simulators

The use of elementary simulation models (such as Markov models) of the vegetation dynamics is a research area that is in a development stage than in a application stage. Nonetheless, the capabilities that these methods provide to aid in the planning of management plans to maintain a given level of diversity is considerable.

### Case 4: Ecosystem/Landscape Management in National Parks in the United States, Canada and Australia.

One set of applications are those of Kessell and Potter and their colleagues (Kessell 1976, Cattelino et al. 1979, Potter et al. 1979, Kessell and Potter 1980, Kessell 1979a&b). These authors developed several models for landscapes in the United States, Canada and Australia for use in landscape management of national Parks in all three countries. The models were based on Markov models with parameters that were adjusted to predict vegetation change at different locations on a landscape. Each model could be used to project change over time intervals of about 10 years for each of the cells in a grid covering an entire park landscape. Output of a model was in the form of a dynamic map of the park's vegetation. This map changed over time as succession, fires and other disturbances changed the landscape it represented. By applying models somewhat like the HEP models described above, it was also possible to produce maps of the potential habitat of several hundred species of plants and animals for given years in the future (Kessell 1976). The latter versions of these models were also programmed so that a land manager could interact to test different management strategies and to see what the consequences might be (Kessell et al. 1982). Intervening conditions such as an unexpected wildfire that altered the vegetation over a part of the park could be included in the basic data base and the effect of this fire could be assessed and new management priorities could be developed. The potential of such tools for developing a landscape management plan is considerable (Franklin 1979). Kessell (pers. comm.) indicates that the implimentation of a conversational and interactive computer program with which the manager could work in a direct fashion and the involvement of the manager in the model development from the start of the project were important elements in the success of the resultant models. The basic models used in the approach are relatively simple and straight-forward but they do require the attention of an expert ecological modeler (or a research team) for their development. Such individuals are in a short supply with respect to the present need.

## Individual-Plant Vegetation Simulators

The advent of computer models that simulate the vegetation dynamics of small patches of a landscape by following the change of the individual plants on the landscape. Although the level of detail in the output of such models is quite large, the models are well within the computational limits of most "main-frame" computers and can in many instances be used on the PC-sized computers that are available at a relatively low cost. The most appealling aspect of these models in terms of the evaluation of habitat change is that the models have demonstrated some capability to predict the response of vegetation to novel situations (eg. climate change or pollutant stress effects, see

Shugart 1984 for some examples and references to a wide range of model tests).

Example Case 5: A Dynamic National Inventory in Sweden

One very impressive application of modern technology to develop a management information system that can be used to develop dynamic national policies for diversity management is the Bonitering system developed by Hagglund and his colleagues in Sweden. In this case, a systems model developed that was capable of projecting the response of the forest to different forest management strategies with considerable detail (Hagglund 1981). The national forest inventory was them augmented by adding a survey of the flora and a number of environmental measurements were also associated with the inventory plots. The protocols for developing the additional data were published in handbooks that were available and understandable by the public (Hagglund and Lundmark 1985 a,b,c). In total over 100 variables including such detailed information as adundance of certain species of mosses, mushrooms, berries and a wide range of wildflowers were measured at each of the study plots used to estimate the national timber reserve. The data set was used to determine the relationships among the biotic variables and the aspects of the forest structure that might be altered by timber utilization. The result of this study was an abilty to predict the regional effect of timber harvest on the seasonal availablity of mushrooms (picking edible mushrooms being a popular recreation during the summer) or on particular species of herbaceous plants. The

Swedish study is unique in its consideration of small plant species in a national survey that can be used to project change over time.

Example Case 6: Theoretical Explorations using Habitat Simulators.

Most applications of detailed models to the simulation of habitat dynamics are currently in a development stage. The field testing of the models is a clearly needed next step. The utility of the models as aids in designing management strategies for the creation of specialized habitats and the use of the models to project the detailed nature of each element of a landscape mosaic over time make them potentially valuable tools in designing management plans to maintain certain species or a given level of species diversity.

Some of the models that are used as the habitat simulators in these methodologies were originally designed to predict timber yields under different harvest strategies (Moeur 1985, Sweeny 1985, Benson et al. 1985, Brand et al. 1985, Holthausen 1985). These models have been tested extensively in some cases with respect to the simulation of vegetation change under different conditions but the level of testing of the parts of the models that simulate the responses of species to the vegetation change are less tested. One valuable aspect of these models is that they typically simulate the timber yield from the management procedures that generate the habitat change. Thus, the output is also useful in understanding the economic trade-offs involved with different strategies of managing productive landscapes.

Other of the detailed vegetation simulators were developed for ecological research (Shugart 1984). These models typically treat the complex mixed-species and mixed-aged aspect of more natural vegetation found in less managed ("more natural") ecosystems. The models are calibrated from the life history attributes of the species of plants making up the vegetation and do not require nearly as much detailed calibration data as the management models. Smith et al. (1981) provides an example of using the FORET model of Appalachian hardwood forest to assess the habitat available to several species of birds following a selective timber harvest of a watershed in Tennessee. In this case, multivariate statistics were used to associate the suitability of the habitat over time (as simulated by the model) for each of the species.

## Discussion

The methods that have been designed to aid in managing landscapes for species diversity are in their present state of development limited in their utility by a need for more fundamental data and by a need for more experience in applying in the techniques. In a sense this is a good situation in that effort put into these approaches will have a relatively high likelihood of producing improved results. One sees this in the cases of the HEP schemes for managing for habitat - the studies that have a longer history and in which a considerable effort has gone into obtaining data for understanding the habitat requirements of a wide suite of species have been the most successful. While it is not surprising that the more information that one has the better one's ability to make good decisions, the need for more information does mask potential difficulties with almost all of these approaches.

We do not know at present what considerations may be involved with the perfection of these techniques other than the obtaining of more and better data. For example, Rice et al. (1985) report a 90% success rate in predicting relative abundance of riparian bird species diversity in Arizona and Rotenberry (1985) used similar methods but was unable to have anything like this success rate for scrub-steppe birds. The interpretation of these findings (Rotenberry 1985) is that the populations of birds scrub-steppe habitats are sufficiently variable to be unpredictable under almost any circumstances. Riparian zones in arid environments may have a more predictable avifauna. It should be expected that in applying habitat management methodologies that the success in predicting diversity may vary widely as a function of the ecosystem that is being considered.

The HEP procedures, as they are currently being implemented, often are viewed by research scientists as being too simplistic. This simplicity has direct appeal to land mamagers who are often confused by the detail of some of the other methods of assessing habitat. There is little in the HEP procedures that would give one much confidence in their abilities to simulate the response of a community under novel or unusual conditions. The procedures are in need of considerable field testing and will probably improve as more information is added to the characterizations of the animal habitats.

The more dynamic approaches to managing animal habitat are still for the most part in the research and development phase and have not been widely applied (with some exceptions that are noted in the earlier sections). An ability to simulate the long term dynamics of habitats over a landscape mosaic is an important tool that will be needed if landscape management for diversity is to become successful over long time periods.

Almost all of the methods that have been described have been developed and applied in the temperate zone in relatively well understood ecosystems and to relatively well known species. How can these methods be applied less well known ecosystems (particularly in the diverse ecosystems of the tropics)? It will require a considerable commitment of funds and effort to accomplish such a goal. The development of habitat simulation models has the desirable spin-off that the capability of projecting landscape change over time is an invaluable asset for land-use planning in general.

## RECOMMENDATIONS

To this point, this report has identified a management technique refered to as ecosystem/landscape management and has attempted to present the origins of the technique and actual applications in several different situations. Ecosystem/landscape management draws heavily on the idea that natural landscapes function as dynamically changing mosaics and on the use of computer models to implement management techniques. If these procedures are to be used and perfected there are several necessary steps that should be taken.

1. There is a need to acquire land. The application of any mosaic concept of land stewardship necessarily requires large tracts of preferably contiguous land. The land does not necessarily need to be pristine in its character although for most ecosystem types such areas would be in short supply at the initiation of the landscape management program. The amount of land needed to maintain a mosaic of habitats supporting a diverse array of viable populations would be the product of the size of a patch and the time that ecological succession typically requires in the system. The minimal size of a habitat patch might be determined using species/area considerations. The length of time needed for succession would determine the rotation time.

2. There is a need for funding in basic scientific studies in the area of landscape and ecosystem ecology. Such an initiative exists in the National Science Foundation in which workshops and symposia among ecologists were used to determine the need for such studies. There is a parallel need in the more applied agencies particularly the National Park Service, USDA Forest Service and The Bureau of Land Management.

3. The development of models that can simulate the dynamics of vegetation with sufficient detail to be used as habitat simulators is needed. Testing protocols for these models should also be developed.

4. Training programs to develop scientists and managers with the skills to develop and apply these programs should be developed supported in Universities. The nature of the training programs should be interdisciplinary. There seems to be a potential shortage of personel trained in the diverse array of

skills needed for the development of these programs.

5. Since much of the global diversity is in tropical areas particularly tropical forests, effort should be made to extend this management technique to the tropics. The landscape-mosaic view of ecosystems may be the most logical methodology to use in tropical areas. One finds this in the statement of Whitmore (1982), an experienced tropical forest ecologist,"... forests throughout the world are fundamentally similar, despite great differences in structural complexity and floristic richness, because processes of forest succession and many of the autecological processes of of tree species are cosmopolitan. There is a basic similarity in space and time because the same processes are at work." The idea of a dynamic landscape mosaic is an idea that both historically and recently has had much currency among tropical ecologists (particularly those involved with forest ecosystems). There is some likelihood that utilization of tropical forests that are not converted to nonforest will be in the form of selective timber harvest and ways to assess and utilize the results of such techniques are emerging (Johns 1985). For all these positive signs, it is quite likely that the best management for diversity in tropical ecosystems (particularly in the diverse tropical moist forests) will be some form of ecosystem management. This is because there are simply too many species in these systems about which we know too little. Of the land management plans that were discussed above some version of the area-diversity plan would seem a logical first step for many systems.

In summary, the use of models and other techniques to project change in ecosystems over time and the development of data sets that will allow active management to create habitat for important species or of species that are indicators of conditions that seem to favor sets of others species are technologically feasible - given work that has developed in ecology to date. There are logical steps involving the testing of theories, application of methods, training of personel that can be undertaken within existing structures inside the United States research support structure. The tropical ecosystems (which are exceedingly diverse in some cases) are the most challenging areas for the application of these technologies.

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Contract Paper No. 4

# A Science of Refuge Design and Management

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# A SCIENCE OF REFUGE DESIGN AND MANAGEMENT

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## ABSTRACT

The last two decades have seen scientific growth in two areas that augurs well for a sound conservation science. First, field observations and experiments have elucidated the nature of ecological communities and ecosystems and have shown that, though these are less holistic than they were often portrayed, nevertheless there are frequently interactions among species and between species and their abiotic environment that are critical to the survival of the interacting species. The same research has suggested management techniques for various ecosystems, communities, and community components. Second, the theory of island biogeography, though not leading directly and deductively to particular refuge design methods as was originally hoped, has focussed interest of the concept of the minimum viable population and the forces that determine this minimum. In so doing, it has spawned research that, together with that on community and ecosystem ecology, should provide sound biological guidelines for how to achieve particular conservation goals.

Key to effective refuge designation and management is an explicit consideration of the goals of the refuge. The usual goal, until recently, has been preservation of single species or a small number of species. However, such goals as maximized species richness, maximized species diversity, maintenance of various forms of community structure, and maintenance or maximization of various ecosystem functions (such as gross production or particular nutrient cycles) have frequently been articulated. The community goals are likely to be technically more difficult to achieve than the species or ecosystem goals, though the same management procedures may serve several different goals. However, there are likely to be situations where the goals conflict; for example, management to maximize population size of a particular species may well lead to decrease species richness of the community, or modified nutrient cycling in the ecosystem. There is no easy resolution of this problem, but there is every reason for optimism that once the goals have been elucidated, and sufficient economic resources committed to the project, technologies exist or can be developed to achieve the goals.

## INTRODUCTION

The discipline now called "refuge design" did not formally exist until ca. 1971. This does not mean that there were no wildlife refuges, national parks, game reserves, etc., or that no thought was given to where to put them and how to manage them, only that there was no codification of principles for establishing and managing them. Even as young and development-oriented a nation as the United States established its first national park in 1872 and its National Park System by 1916 (Stone 1965), while in Europe and Asia hunting reserves of aristocracy achieved one of the main goals of national parks - maintenance of wildlife - centuries earlier.

At first the principles of locating and operating such refuges seemed so obvious as to remain unstated. Every species has a characteristic range of habitats in which it will thrive, and some have other species, such as mutualists, food plants, pollinators, or hosts, that are necessary for long-term survival. Most early conservation efforts were designed to save particular species of special interest, such as endangered species, endemic species, or simply spectacular species. In the United States, for example, refuges were established specifically for the California condor (*Gymnogyps californianus*), saguaro cactus (*Carnegia gigantea*), Joshua tree (*Yucca brevifolia*), Kirtland's warbler (*Dendroica kirtlandii*), and other species.

It usually seemed a straightforward matter where to locate such reserves: where the species is found. For a narrowly endemic species, there may be little choice in the matter. The saguaro, for example, is found in the United States only in parts of southern Arizona (Gleason and Cronquist 1964). For a species more widely distributed, especially if it is common only at certain sites within its geographic range, one would attempt to specify its preferred physical habitat, and insure that the refuge had a substantial amount of it. Joshua tree, for instance, is found widely in the northern and western parts of the Sonoran floristic province, but characteristically grows only at the upper edge of a plant community dominated by creosote bush (*Larrea divaricata*), often forming a fringe about a mile wide where cool mountain slope desert abuts warmer low-elevation desert (Gleason and Cronquist 1964). There may also be an obvious necessity for other species if a target species is to be conserved. Joshua tree is pollinated by moths of the genus *Pronuba*, and propagation of the tree would be impossible without the moth. Often an obligatory interactant would automatically be maintained in any refuge for the target species, but this is not mandatory. Temple (1977; *cf.* Owadally 1979) suggests such a scenario for the decline of the endemic sapotaceous tree *Calvaria major* on the island of Mauritius. Though this species was formerly very common, by 1973 there were only 13 dying individuals, each thought to be over 300 years old, in protected native remnant forests. The trees produce fertile seeds, but these never germinate in nature, and it is possible that they required the extinct dodo, *Raphus cucullatus*, to abrade them and pass them through its gut.

In many cases the habitat requirements and key interacting species could be deduced from rather straightforward, short-term observational research, so it was apparent where to establish and how to manage a refuge. Other times, however, habitat requirements and important species interactions may be remarkably subtle, and laborious field research may be necessary to understand them. Furthermore, there may be surprising concatenated effects that ripple through a chain of several species when one of them is disturbed.

For example, the endangered red-cockaded woodpecker, *Dendrocopus borealis*, requires moribund trees of longleaf pine (*Pinus palustris*) and loblolly pine (*Pinus taeda*) for nesting. Longleaf pine, in turn, cannot persist without occasional fires that destroy seedlings of other species that would otherwise outcompete it (Schiff 1962). Such fires are greatly facilitated by wiregrass (*Aristida stricta*), and it is difficult to maintain longleaf pine in the absence of wiregrass. Wiregrass propagates almost exclusively vegetatively and not by seed, so when a previously cleared area is left fallow, it is unlikely to revert to wiregrass even if wiregrass was the original ground cover. Conservation of red-cockaded woodpeckers in at least part of their range (north Florida) thus requires maintenance of wiregrass and entails conserving an entire community of species adapted to frequent fires.

Not all species are so inextricably intertwined with so many other species as the red-cockaded woodpecker is, but many have at least a few obligatory interactants, so that to some extent a refuge must be designed for at least a few species, not just one. This realization is not completely new, but the emphasis on community, rather than species, in conservation has become progressively stronger during the last decade (a point I will return to below). The research that elucidated the longleaf pine-wiregrass-red-cockaded woodpecker connection was intensive (*cf.* Stoddard 1936, Thompson 1971, Christensen 1981, Wood 1983), but no shortcut would have enabled us to design a suitable reserve. However, insightful and arduous field study can determine enough about the habitat requirements and interacting species of any target species to allow the beginning of scientific refuge design and long-term conservation.

Until the last decade, this was virtually the entire thrust of scientific conservation: In the sections on site selection in standard conservation texts of ten years ago (e.g., Cox 1974, Owen 1975) it is taken for granted that the key to conservation is the autecology of species of interest, and then conserving enough of their habitat. Not much attention was paid to exactly how much habitat is "enough," which is surprising since this is an obvious question and was even asked during this period by the current President of the United States (Segerberg 1971). Sometimes subsequent management would be needed after site selection. Primarily this would be protection, though there were certain recognized methods for maintaining particular habitats. There were no general guidelines beyond the naturalist's feel for what would be a sufficient size and optimal shape for a site.

However, sometimes more than just designating, acquiring, and then protecting a site is necessary. For example, without occasional fires, longleaf pine, wiregrass, and the species associated with them are replaced in a natural succession by less fire-tolerant species that are otherwise competitively superior. In north Florida two centuries ago, such fires were started every few years by lightning. The longleaf pine community spread quite uninterrupted for tens or hundreds of kilometers, so any particular part of the forest would certainly burn every few years. Two developments combined to change this scenario. First, for at least the last 150 years, north Florida has undergone progressive "insularization" of the habitat as expanding agriculture turns the landscape into a patchwork of fields and forests of various sorts. Undisturbed forest, dominated by longleaf pine, has been increasingly restricted to fewer, smaller, and more isolated "islands" in a sea of fields. Similar insularization has occurred over much of North America, Europe, New Zealand, and even the tropics (Burgess and Sharp 1981). In longleaf pine forests, the isolation of separated islands would suffice to prevent natural fires that start in one area from spreading to other areas, and thus would decrease the frequency of fire in any one island, possibly to the point where succession would ensue and a hardwood forest would replace the pines.

Second, fire seems to arouse human fear. To amateur and even some professional naturalists, especially in Europe and North America, fire has usually been anathema. Many "primitive" peoples use fire on a routine basis to maintain particular habitats, and it has been known for many years that numerous vegetation communities around the world are maintained by occasional fires (e.g., Garren 1943, Shantz 1947). But the spectacle of burning trees and, occasionally, scorched wildlife seems to inspire visions of horrible suffering and economic disaster. This view of fire makes it very difficult, on pyschological grounds alone, to use controlled fire as a management tool to maintain a habitat. In the United States, for example, a national campaign to eliminate natural forest fires peaked in the 1920's with passage of the Clarke-McNary Act providing federal assistance to states with fire prevention programs. The U.S. Forest Service virtually prohibited controlled burning at this time, despite a century of study showing that longleaf pine requires fire (Schiff 1962). Managed refuges were not immune to this pressure: for example, the United States National Park Service has been hostile to the use of fire for habitat maintenance for much of its history (Hendrickson 1973), while in Germany there has been a prohibition on burning in nature reserves and landscape-controlled areas since about 1924 (Makowski 1974).

By 1936, Stoddard (1936) and others had shown how important fire is to maintenance of certain natural communities, especially those with longleaf pine (cf. Garren 1943, Schantz 1947, Schiff 1962), and by the 1940's the U.S. Forest Service no longer demanded complete proscription of fire. But it was not until a report by a committee of the U.S. National Research Council (Leopold 1963) that use of fire in wildlife and habitat maintenance began to make headway in the U.S. National Park Service. However, burning in a way that mimics nature is no trivial matter, especially when fire has been excluded and fuel has built up on the ground. Even with recent great improvement in the technology of controlled burning (Tall Timbers 1976, Butts 1985), the insularization of habitat described above makes the process expensive. Starting and controlling fires in myriad small refuges is vastly more difficult and costly than the same task would be in one large refuge. A Florida refuge manager has suggested that it is impractical to perform controlled burning on plots of less than about 10 ha. So part of the answer to the question of how much habitat is enough is linked to the question of what management is required.

Although we are more often interested in maintaining climax communities than earlier stages or disclimaxes (like the fire disclimax in north Florida), even climax communities usually require maintenance, or at least protection. And there is increasing realization that there are non-climax communities of conservation interest (Stone 1965, Pickett and Thompson 1978, Gilbert 1980) that can be maintained only if refuges are actively managed.

There seem, then, to be three components to a science of refuge design that would replace the rather intuitive approach that conservationists originally used. At the outset must clearly be stated what the goals of the refuge are: To conserve particular species? Particular communities? Entire ecosystems? As many species as possible? Particular habitats? First, once the target (species, community, or ecosystem) is designated, how does one determine how large a site, or how large a population, is enough for conservation purposes? Why, in fact, is one redwood tree not enough? Second, one must ask whether it is even reasonable to focus on single species or even small groups of them for conservation. Are communities such holistic entities, with such tight and organic interrelationships among the component species that it would be impossible to save only some of them? Third, what kinds of management are needed? Is it possible simply to set aside a refuge and protect it? If further intervention is needed, how do we determine what exactly to do?

All three of these questions have motivated intensive research efforts in the last decade or two, so that there are certainly in place all of the prerequisites for a science of refuge design and management, though it is only very recently that the necessary connections are being drawn among these efforts to produce a unified science. In the remainder of this essay I will sketch the history and status of thinking on these three topics.

## THE MINIMUM VIABLE POPULATION CONCEPT

Although it has long been recognized that small populations are fragile for several reasons (particularly genetic ones), the publication in 1967 of The Theory of Island Biogeography by Robert H. MacArthur and Edward O. Wilson led to a more formal consideration of this fragility. Broadly speaking, there are four reasons why very small populations are unlikely to persist (Shaffer 1981, Soulé 1983). Consideration of these reasons has led to the concept of the "minimum viable population" (MVP), that number of individuals such that when populations fall below this point they are doomed to quick extinction.

MacArthur and Wilson (1967) originally conceived of this point as a clear inflection in a curve relating expected time to extinction to present population size (Fig. 1). Populations above this point were virtually immune to extinction, while those below this point were likely to go extinct very quickly. However, MacArthur and Wilson (1967) treated only one of the four reasons for the increased probability of extinction, and their simple model has been made increasingly realistic (Richter-Dyn and Goel 1972, Shaffer and Samson 1985) for the one force that they included, with the result that for this force one might expect a more gradually increasing expected persistence time as population size increases (Fig. 1). For the other forces the exact shape of the curve has not even been predicted, and there are no empirical data. However, we are concerned not so much with the exact shape of the curve as with the proposition that swift extinction is ensured for very small populations, so the MVP can serve as a metaphor for this proposition, even if we cannot specify a specific number as the critical size.

### i) Demographic Stochasticity

The one force treated by MacArthur and Wilson (1967) is demographic stochasticity, the random variation in population variables such as sex ratio, birth and death rates, or the distribution of individuals among age classes (May 1973). This force is more threatening to a small population. For example, if N is the expected number of offspring in any generation, the probability that they will all be of one sex is $2^{1-N}$. Thus extinction from this source is vastly greater for small than for large populations. There is also typically an increasing skewness towards males from a source discussed below, inbreeding depression (Senner 1980). Furthermore, the skewed sex ratio increases the rate of inbreeding.

ii) Genetic Deterioration

Conservation biologists recognized early that various stochastic genetic factors impose a limit on how small a population can be and still persist. The main such genetic factor is inbreeding depression. Most peoples have incest taboos, and it is often said (*e.g.*, Wilson 1978) that these evolve, culturally and/or genetically, in response to the observation that harmful biological effects increase when near relatives mate. Whatever the actual basis for incest taboos, animal breeders have recognized for several centuries that continued inbreeding within a herd is associated with deterioration (Darwin 1868, Wright 1977). Similarly, among plant breeders, deleterious effects of inbreeding have long been recognized (*e.g.*, Koelreuter 1776, Gaertner 1849, Darwin 1876). The formal study of the genetics of inbreeding was initiated in 1906 (Wright 1977), and sufficient research has since been performed that Ralls and Ballou (1983) conclude that inbreeding depression is likely for animals forced to breed within small groups. Inbreeding depression has two aspects:

1) The increased frequency of diseases caused by recessive alleles, of one or a few genes, that in homozygous condition are

severely debilitating. A common congenital diaphragm defect in the endangered Brazilian golden lion tamarin (Leontopithecus rosalia) appears to be such a disease (Bush et al. 1980). There are fewer than 200 individuals of this species in captivity and 30 in nature (Pinder and Barkham 1978), so inbreeding is increased and has probably increased the frequency of this disease.

2) A more important aspect of inbreeding depression is heterosis: the more genes for which an individual is heterozygous, the more vigorous and fertile, on average, it is likely to be. This is true even if there is no obvious disease, like the diaphragm defect just mentioned, present among less heterozygous individuals. Since inbreeding decreases average heterozygosity, and decreases it faster the smaller the population size (Hartl 1981), decrease in vigor and fertility is a common effect of inbreeding. As Ralls and Ballou (1983) point out, since such effects are subtle, this consequence of inbreeding could be confirmed only by detailed studies and meticulously kept pedigree records, such as those now available for some zoo animals.

For plants, there is less evidence for inbreeding depression, but there is good indication that in at least certain species inbred populations with lower heterozygosity suffer from decreased fertility as well as size and other attributes normally associated with fitness (Schaal and Levin 1976, Clegg and Brown 1983, Ledig et al. 1983).

In the wildlife management literature, even in the face of evidence of at least the possibility of inbreeding in small populations such as those on refuges, there is occasional scepticism about its importance. For example, Whitehead (1980) states that "adverse comment about the ill-effects of inbreeding is . . . often exaggerated." Such scepticism, though overstated, may not be completely unwarranted. That is, one cannot automatically expect inbreeding depression to threaten every single small population of every species. Nevertheless, though its exact importance for particular species remains controversial, the dangers of inbreeding depression in small populations were well known to early conservationists. Gross (1928), discussing the failure of a refuge to save the declining population of heath hens (Tympanuchus cupido attwateri) on an island off Massachusetts, said, "The problem of saving the Heath Hen is not the simple one of providing protection against hawks and cats and supplying food when needed, but is much more complex. There are other factors such as the inadaptability of the species, excessive inbreeding, excess male ratio and disease which are most important in the decline of the Heath Hen."

The role of inbreeding depression in extinction is substantiated by several other examples. It is implicated in the elimination of the ivory-billed woodpecker (Campephilus principalis) (Allen and Kellogg 1937), while Frankel and Soulé (1981) cite many laboratory studies in which the great majority of inbred lines went extinct. For example, of twenty lines of house mouse (Mus musculus) subjected to inbreeding, only ten survived to the fifth generation and only one to the twelfth generation (Bowman and Falconer 1960). Since many of the organisms, such as guinea pigs and chickens, subjected to such experiments were already inbred and thus might have been expected to have been purged of alleles that would cause major inbreeding depression, such experiments constitute strong evidence that inbreeding depression can play a proximate role in extinction. Many more informal breeding schemes for domestic animals that have run afoul of fertility and viability decrease after inbreeding (Soulè 1980) lead one to the same conclusion.

At exactly what population size inbreeding depression becomes a threat to population survival depends very much on the species - its evolutionary history, normal breeding system, etc. If a species typically inbreeds, it is less likely to be greatly affected by additional inbreeding imposed by a decreased population. Those alleles that would be severely debilitating when homozygous are more likely to have already been selected out of the population. Animal breeders, who typically work with already selected lines, have found empirically that a per generation rate of inbreeding exceeding 2 or 3% allows deleterious alleles to be fixed more rapidly than selection can eliminate them (Soulè 1980). Soulè (1980) suggests that for conservation of wild, non-inbred animals, 1% would be a better limit, and this translates to an "effective population size," $N_e$ (see below) of 50 individuals. Breeders have also found empirically that there is usually a major effect of fecundity as the inbreeding coefficient approaches 0.5 - 0.6, and Soulè shows that this figure suggests that the expected number of generations to a point where rapid extinction is insured is about $1.5 \times N_e$.

Another aspect of genetic deterioration is genetic drift, which will lead to loss of alleles. To understand the effect of genetic drift, one must grasp the notion of "effective population size." Alleles will be lost from a population of size N at a rate roughly proportional to 1/2N only if there are equal numbers of both sexes, mating is random, and the population size stays the same from generation to generation. Any deviation from these conditions causes greater loss of alleles. For any particular population of size N, the effective population size $N_e$ is defined as the size of an unchanging population with sex ratio 1:1 in which mating is random and which loses alleles by drift at the same rate as the particular population. This definition has two consequences. First, the effective population size is often vastly smaller than the censused population size (e.g., Hedrick 1983). Secondly, loss of genetic variation from drift in any population will be approximately at the rate of $1/2N_e$. One sees, then, that this loss will be at a much higher rate in smaller than in larger populations, and may be at a much higher rate than one might have expected simply from counting the individuals in the population.

There is dispute about just how small a population must be in order for drift to be a significant force in removing alleles in the face of normal natural selection, with estimates ranging from $N_e = 50$ (Berry 1971) to $N_e = 500$ (Franklin 1980). However, extinction of small populations seems frequently to take place so quickly that one might not have expected evolution to have solved the problem, even if there were a wealth of genetic variability (Simberloff 1985a). In other words, it is likely that other factors will eliminate a small population before failure to evolve endangers it.

### iii) Social Dysfunction

Soulé (1983) observes that some species have characteristic social behavior that renders them more liable to extinction. The increased risk arises in two ways. First, elaborate group mating displays, such as the lekking behavior of the heath hen (Bent 1932), make it far easier for certain other forces, such as hunting, to make enormous inroads. Other kinds of social behavior, such as group defense and herding or schooling, can have a similar effect.

Second, once a population is already small, there may simply be too few individuals to stimulate or consummate critical behavior. A good example is the "Fraser Darling" effect by which certain species satiate their predators by synchronous breeding. One proximate mechanism by which colonial birds achieve such synchrony is group stimulation of ovarian development of the colony's females (Daly and Wilson 1978, Wittenberger 1981). There are similar examples of social stimuli producing reproductive synchrony in mammals (including human females (Daly and Wilson 1978)), though there is as yet no evidence that predator-satiation is the selective agent. One can easily imagine, then, at least for birds, that even if the distribution of clutch sizes remained constant as population sizes decreased, mortality from predation would increase.

Asynchrony could also reduce mean clutch size. A species may require some sort of social facilitation in order for mating and offspring production to be vigorous (Halliday 1978). Asynchrony between the sexes could even prevent mating entirely. Allen (1934) argues that some bird species have short oestrous cycles during which fertilization must occur, while males have short intervals during which they can fertilize a female. If there are too few individuals in a population, the probability that a fertile female encounters a capable male may fall precipitously.

### iv) Extrinsic Forces

Temporal variation in habitat parameters and the population sizes of enemies as well as random catastrophes such as fires, storms, and epizootics can lead to extinction. Mortality caused by these forces is density-independent. If the per capita probability of death is D, and if deaths of individuals are independent, then the probability that all N individuals in the population die is $D^N$, an exponentially decreasing function of N. Probably the spatial extent of the population, or the number of separate populations of the species, is as critical a factor as the number of individuals for many extrinsic forces. For example, the fire that ravaged the last population of the heath hen would not have wrought such havoc had there been several populations, even if all of them were smaller than the single one on Martha's Vineyard (Drury 1974). Similarly, the epizootic disease, blackhead, that decimated the same population would have been restricted had there been discrete populations.

Whatever minimum viable population size is deduced from a consideration of forces (i) - (iv) must, for conservation purposes, be translated into an area. That is, assuming that habitat were optimal, one wishes to know how much of it must be set aside. Once the group of species whose conservation is desired (or required, if a particular target species has obligatory interactants) is designated, the species whose MVP requires the largest area determines the minimum area for a refuge. That is, the smallest possible area for the desired refuge will be the biggest area required by any single species. Such species will likely be large vertebrates, especially predators, and their minimum areas will require the same sort of detailed field study discussed above in connection with elucidating community structure. One will need to know the behavior, resources requirements, and other aspects of a species' biology in order to extrapolate to an area requirement. For vertebrates this area may be very large indeed. An early effort of this sort (Allen and Kellogg 1937) concluded that a single pair of ivory-billed woodpeckers would require about 3,800 acres of prime habitat. If the MVP were 50 pairs, 300 square miles would be needed. However, neutral genetic variation would be lost from a population of 50 pairs in perhaps 100 generations, so this is not a realistic long-term MVP.

The above four reasons for enhanced danger of extinction for small populations omits the single largest force causing extinction today: human activity (Simberloff 1985a). In fact, the vast majority of documented extinctions and endangered populations were ultimtely brought about by a variety of anthropogenous forces (e.g., Ziswiler 1967). Assigning a cause to extinction is often misleading or impossible, just like attributing a death to a particular disease. Many cancer patients, for example, officially die of congestive heart failure, though the ravages of cancer led directly to the heart condition. By the same token, habitat destruction, hunting, and all the other human activities that lower population sizes and restrict ranges can almost always be viewed as the ultimate cause of extinction, for they reduce populations to the small size at which the above four factors automatically come into play (Simberloff 1985a).

A series of papers (e.g., Diamond 1975, Wilson and Willis 1975) proposed that the theory of island biogeography (MacArthur and Wilson 1967), in addition to suggesting increased study of the fragility of small populations, mandated a particular kind of refuge design, namely single large, round refuges rather than long, thin refuges of equal area or groups of small refuges of equal total area. Since the theory of island biogeography deals only in numbers of species, not their identities, these suggestions would in any event apply only if the goal for a refuge were maintenance of highest possible diversity.

However, it was quickly demonstrated (Simberloff and Abele 1976) that both of these suggestions are empirical propositions, and do not derive from the theory. Only now, a decade later, is it generally agreed among conservationists that the theory does not generate these predictions (Soulé and Simberloff 1985), and that they are wholly an empirical matter.

Both experimental studies and examination of data in the literature have now focussed on this matter, and the general result is that, though there are specific exceptions in each case, most of the time the results are exactly the opposite to the original suggestions: groups of small sites tend to have more species than single large sites and long thin sites tend to have more species than round ones (Simberloff and Abele 1982, Simberloff and Gotelli 1984, Blouin and Connor 1985, Quinn _et_ _al._ 1985). However, these rules of thumb have extremely limited utility (Soulé and Simberloff 1985) for three essential reasons:

1) Minimum viable population sizes would put a lower limit on the sizes of the small sites in a group of small sites. As noted above, these sizes are taxon-specific. Similarly, a site cannot be too narrow or else it will cease to function as the habitat of interest and instead comprise an ecotone of very different character (Simberloff 1985b).

2) The cited studies are based only on the criterion of species richness, whereas one is almost certainly going to be interested at least to some extent in matters of species identity. There may be special concern for rare species, vulnerable species, species useful in some way, etc.

3) As a practical matter, conservationists are rarely if ever faced with the decision of choosing one large or several small sites, or of choosing a round or a thin site, *ceteris paribus*. Often there is no choice at all, and one either sets aside a site or does not. In Europe, for example, there is very little land that could conceivably be set aside that does not already have conservation status. In much of the developing world, there are so many other pressures on pristine land that conservationists can at best hope for what is left after other demands have been assuaged. And even when there is a choice, all other factors (such as habitat, cost, vulnerability) are exceedingly unlikely to be equal. The inequalities are likely to be so pronounced that these considerations will rightfully be paramount in conservation decisions.

## THE CONCEPTS OF ECOLOGICAL COMMUNITY AND ECOSYSTEM

The notion that there is an inherent balance of nature traces back to the pre-Socratic Greeks. Herodotus in the fifth century B.C. described mutualistic relationships among species, such as a plover cleaning crocodiles on the Nile. In the next century, Plato's Timaeus myth introduced the metaphor of the superorganism, in which all parts of the universe are viewed as the "organs" of a superbeing. This metaphor suggests that the balance of biotic nature is an aspect of a higher level balance or integrity, and that all species fit together in a harmonious, organized whole. Egerton (1973) describes the trajectory of the balance-of-nature concept, and of its superorganismic metaphor, through the nineteenth century and shows that it has been a dominant theme in Western thought, though it has taken various forms.

Although the Greeks and Romans sought endogenous, homeostatic forces, such as properties of particular species and interactions among species, that kept nature in balance, during the Middle Ages the balance of nature was conceived as maintained by the constant intervention of the hand of God. Beginning with the Reformation there was renewed interest in endogenous mechanisms that would automatically maintain order in nature, though many conceptions of a balance of nature had religious overtones through the nineteenth century.

Through the ages many proponents of the idea of a balance of nature invoked holistic images of ecological communities; sometimes the influence of the Platonic superorganism was quite explicit. For example, Alexander Pope's Essay on Man (1733) included the suggestion that all species are so interdependent that the extinction of any one endangers all of nature. In 1887, the American limnologist S.A. Forbes gave an influential address entitled "The Lake as Microcosm" that invoked the superorganism as a metaphor for a lacustrine community. Such views imply that conservation of a single species is impossible, but the modern idea of ecological community was not fully articulated until later. There were occasional skeptics about whether nature is balanced. Alfred Russel Wallace was particularly ahead of his time and asked, in current parlance, whether the idea of a balance of nature is falsifiable (Egerton 1973). However, the idea that there is an evident balance in nature remained dominant, and even today, though rarely articulated in scientific discourse, can be seen to be implicit in some ecological models and is often explicit in the literature of the environmental movement (Simberloff 1980, 1984a).

Egerton (1973) views Forbes' discussion of the lake community and

Mobius's discussion in the same year (1887) of an oyster community as the beginning of a scientific conception of an ecological community, but the signal work, which set the stage for half a century of ecological research, was the first American textbook of ecology, _Research Methods in Ecology_, by Frederic Clements (1905). In this work Clements forcefully described plant communities as integrated superorganisms. This conception of nature as organized into communities became the dominant paradigm not only in America but in Europe as well (McIntosh 1980, Simberloff 1980), to the extent that observations on individual species' populations, independently of the community, were not viewed as acceptable scientific research, and H.A. Gleason and others who suggested an alternative, individualistic view of ecology, focussed on single species populations, were academically ostracized (McIntosh 1975).

McIntosh (1975, 1976) demonstrates how dominant this paradigm was until recently. One can cite Shelford (1913), a leading animal ecologist in the first half of this century:

> "Ecology is the science of communities. A study of the relations of a single species to the environment conceived without reference to communities and, in the end, unrelated to the natural phenomena of its habitat and community . . . is not properly included in the field of ecology."

The modern superorganismic community concept arose primarily among plant ecologists, but it quickly won adherents among zoologists and limnologists. Allee by 1931 had aligned his work on animal symbioses with the superorganismic paradigm, observing ". . . a more or less characteristic set of animals which are not mere accidental assemblages but are integrated communities." Emerson's similar focus led him to the notion of an "integrated ecological community" (1939). Thienemann described a lake community as "a unity so closed in itself that it must be called an organism of the highest order" (McIntosh 1975). Probably even more important than the zoological and limnological support in establishing the superorganism paradigm was Elton's work (1927) on food chains as pathways for community energy flow. Although Elton himself was not an adherent of the superorganism view, his discovery provided such a diagrammatic analogy to the physiology of an individual organism that it was quickly assimilated as an integral part of the superorganism, in fact, one of the main forces giving it organismic cohesion (Simberloff 1980).

The power of the superorganism paradigm was sometimes sufficient to preclude gathering data on individual species, just as Shelford would have wished. Margaret B. Davis, in her studies of long-term vegetation change, has been frustrated by this lacuna: "We do not know what the virgin vegetation of the pioneer days was like because all the ecologists were so busy looking for a non-existent climax [community] that they forgot to record what was actually growing there" (Colinvaux 1973).

In spite of the antipathy of students of the superorganism towards population-level research, there was a gradual realization that many field data on communities simply did not accord with the idea of a superorganism. The watershed year for the overthrow of the superorganism paradigm was 1947 (Simberloff 1980), when three respected plant ecologists independently published data-based papers in Ecological Monographs forcefully attacking the Clementsian paradigm. They all cited Gleason's individualistic conception of plant communities as the correct alternative. Subsequent research on plant communities along these lines, especially by Curtis (1955, 1959) and Whittaker (1956, 1967), led to the replacement of the superorganism, so that the new generation of ecology textbooks that exploded on the scene in the early 1970's (Colinvaux 1973, Collier _et al._ 1973, Krebs 1972, Poole 1974, etc.) all agreed that species more frequently act rather independently of one another, or at most are coordinated into small groups, rather than into all-encompasssing, integrated communities. The emergence of a unified, quantitative population ecology in the 1960's aided this paradigm shift (Simberloff 1980).

The superorganism is not quite dead - it crops up in the literature of popular environmentalism (_e.g._, Commoner 1971) and in theoretical community ecology (references in Simberloff 1980). Simberloff (1980, 1984a) argues that the vitality of the conception of nature as a delicate balance of populations within an integrated superorganism is a manifestation of Platonic idealism and of psychological unease with the idea of a nature that is not carefully regulated. Whatever the reasons for its persistence, the superorganism concept does not tell us what to do to conserve particular species or communities, nor has it inspired the spate of field studies that _have_ recently provided quite specific guidance in particular cases and a pattern for more general conservation efforts.

All of this is not to say that communities are nothing but the collection of species present in one place at one time. Recent elegant field research, often involving manipulative experiment, has shown that there are embedded in most communities particular pairs or larger groups of species that interact in stylized ways, with two consequences: 1) Certain species cannot be maintained without other particular interacting species. 2) A species may so heavily affect so many other species in a community that the community cannot remain as a recognizable entity without the species.

Examples of pairwise interactions between species, such that their fates are inextricably intertwined, are numerous. I have already discussed the dependency of red-cockaded woodpeckers on longleaf pine and, ultimately, wiregrass. Mutualisms, such as those that tie together an alga and a fungus to form a lichen, are well known. Animals that pollinate particular plants and/or disperse their seeds are another example. Howe (1977, 1983) has shown that several tropical fruit-eating bird species, during certain seasons, feed almost exclusively on fruits of one tree species that, in turn, requires the bird to disperse its seeds. Lovejoy (1973) describes the Brazil nut tree (*Bertholletia excelsa*) as having a single bee species as cross-pollinator, so that monoculture plantations of Brazil nut are impossible. The bee requires other sources of nectar when *Bertholletia* is not in flower. It is quite possible that extinctions of tropical species have already led to cascades of subsequent extinctions but that we have not known about these because the biota was so poorly known (Simberloff 1984b). These interactions can be surprising, and the effects of removing an interactant may not be manifested for a long time, as in the *Calvaria*-dodo example cited above.

Species whose existence is key to entire communities may be called "keystone species" (Paine 1974). The dominant tree species of some temperate forests are such integral features of their respective communities that the communities are often named for them - "oak-hickory forest," "eucalyptus forest," etc. Because trees are so large they generally give physical structure to the community and it seems obvious that they might affect many other inhabitants. Animals may be equally critical to a community, but may be sufficiently small and/or uncommon that one cannot guess at their key roles without intensive familiarity with the system. For example, Paine (1974) showed that the predatory starfish *Pisaster ochraceus* ultimately determines the structure and species composition of rocky intertidal shore communities in the Pacific northwest. To do this he had to remove the starfish, whereupon mussels increased and locally eliminated many other species that could not compete with mussels for space. The entire community was changed when *Pisaster* was eliminated.

Even relatively rare species can play keystone roles, and can have surprising effects on surprising types of species. A good example comes from an extended study, whose publication is in preparation, by Barbara Zimmerman of the World Wildlife Fund. She has discovered in the Amazonian rain forest that wallows of the peccary (*Tayassu pecari*) maintain an entire aquatic community. This community includes 11 species of frogs, of which four breed only in such sites. Several of these frog species are relatively rare, and Zimmerman has constructed artificial "pig wallows" and found that much the same collection of frogs will breed in them. One sees, then, that the peccary is essential to the continued existence of frogs, unless management for peccary-type pools can serve as a substitute.

The research elucidating the key roles played by the starfish and the peccary exemplify the intensive field study that allows critical interactions and physical requirements to be deduced, and there is a growing corpus of literature on such efforts. This corpus suggests that in general nature will be subtle, it will often be interactive, but that interactions for most species will be with a limited set of other species (Paine 1980). This literature has also provided a framework for experimental methods that are useful in understanding particular communities and their species. It is clear that communities are idiosyncratic entities, and that each community of interest will bear study in its own right, but that certain methods and patterns are likely to be common for most communities of a particular broad type (e.g., rain forest, rocky intertidal).

The conception of nature as organized into even higher level entities than communities, namely ecosystems, is a rather recent one. The role of the physical environment in maintaining a balance of nature was quite secondary to either divine intervention or inherent properties of species. To be sure, as early as Liebig's Law of the Minimum (1840) the potential limiting rule of nutrients was noted, and the 19th century plant biogeographers, beginning with De Candolle (1820), recognized the crucial roles of physical factors such as temperature, moisture, and soil type and establishing range limits of particular species and even communities. However, the integration of the physical environment into a dynamic, holistic entity was a 20th century achievement (McIntosh 1980). The key empirical studies were by Elton (1927) and Lindemann (1942), both of whom described the flow of energy through trophic interactions as a unifying ecological principle. Above I have pointed out how the patent analogy of energy flow to individual circulation tended to strengthen the superorganismic concept of the community. A second recognition was that nutrients flowed along with the energy. Nutrient cycles have abiotic components as well as biotic ones (weathering of rock, for example), and it was inevitable that a focus on the flow of nutrients would entail not only a community or group of them but the physical environment in which they are embedded.

Development of the ecosystem concept, by virtue of the size and complexity of ecosystems, received impetus from the mathematical techniques of systems analysis, particularly information theory and general systems theory. McIntosh (1980) sees these elements converging in the 1960's in the work of R. Margalef, E. Odum, and H.T. Odum. Odum (1969) came to conceive of ecosystems as developing in orderly and predictable fashion and culminating in stable entities with biomass and population interactions maximized as a function of energy flow. The plethora of research initiated in the mid 1960's with the International Biological Program (IBP) has led to a number of empirical findings on the functioning of particular ecosystems and to a variety of theories (many of them summarized by Johnson (1977)).

As with developing theories on the nature of ecological communities, the ecosystem theories produced so far do not seem to be specific enough or general enough to lead to rules for erecting maximally effective refuges. However, as with the emprical work on communities, that on ecosystems has demonstrated how ecosystem function can severely constrain conservation efforts and management techniques. For example, Amazonian rain forest, though extremely productive, grows on extremely nutrient-poor soil (Jordan and Herrera 1981). The nutrients that drive the system are maintained in the vegetation itself, rather than the substrate, and the community has many biological mechanisms, such as an absorptive root mat (Jordan 1982) and an extraordinary microorganismic and fungal component (Janos 1983), that prevent nutrients from leaching out of the system. Nutrient cycles are thus much more biotically based and complex than in the average temperate forest ecosystem, and the maintenance of particular tropical forest tree species is impossible without maintenance of the associated mechanisms - the root mat, the mycorrhizal fungi, etc.

Focus on nutrient flow and energy cycling suggests certain techniques for refuge establishment and maintenance. For example, Jordan (1982) has shown that tropical forests can be maintained in the face of limited harvesting, so long as the harvested areas are arranged in narrow strips parallel to elevational contours. This configuration succeeds because it maintains the biological community that minimizes nutrient loss and also prevents rainfall from causing unbearable leaching. There is also the promise that nutrient flow in many ecosystems can be managed even with the loss or depletion of particular key species if other species can be found (or genetically engineered) to serve the threatened function. For example, there is intensive current interest in genetically engineering plants to fix their own nitrogen, and such a development could lessen an entire ecosystem's dependence on particular nitrogen-fixing bacteria and mycorrhizal fungi. It is increasingly apparent, then, that one must clearly delineate the intended function of a refuge - what exactly is it that one wants to conserve?

As with communities, the research elucidating ecosystem structure and function is intensive and suggests that there will be subtle idiosyncratic differences among ecosystems. Certain generalizations are possible - for example, diverse tropical hardwood forests usually grow on nutrient-poor soil. Furthermore, there is a developing corpus of literature on methods of studying ecosystem function (*e.g.*, Odum 1971).

A key concern to both academic ecologists and conservationists is why certain communities and ecosystems are stable and others are not. Why is it that certain systems cease to function and disintegrate even when the imposed stress seems relatively minor, such as addition of a pollutant or harvesting of one population? Part of the confusion surrounding this question stems from the variety of meanings attached to "stability," (Krebs 1985). For example, "stability" might be viewed as a community concept, and an index of stability might then be whether species composition (and relative proportions) change greatly when a given stress is imposed. "Stability" might also be construed as referring to ecosystem function, and a stable ecosystem might then be one in which energy flows and nutrient cycles are relatively little changed by various stresses, even if species composition were completely altered. One sees immediately that, to some extent, the same considerations that go into defining the goal of a particular conservation effort contribute to a definition of stability.

Possibly because the concept of community has an older history than that of ecosystem, most interest has focussed on community stability, and species composition, however indexed, has usually been the attribute of main concern. A venerable tenet of ecology is that species richness is the main attribute conferring stability on a community. The first synthetic statement to this effect was by Elton (1958), and virtually all ecology textbooks of the next two decades espoused this view. Elton's argument rested primarily on two sorts of evidence. First was the observation that certain particular communities, such as island or agricultural communities, that are obviously depauperate in species are notoriously unstable. For example, the devastatation wrought on island communities by invading species are legendary (Elton 1958), and the pest outbreaks of agriculture are widely known (Clark _et al._ 1967). Second was the analogy of a narrow column of bricks to a pyramid of bricks, developed more fully by Wilson and Bossert (1971). The column represented a straight food chain, totalling few species, while the pyramid represented a complex food web of many species. If one removes a single brick from the column, the entire edific tumbles, while a single brick removed from a pyramid is likely to have little effect. In metaphorical terms, in a pyramidal food web each species is buffered from effects of any single other species by its interactions with a variety of other species. This analogy, by explicitly introducing trophic interaction structure as a consideration, subsumes community stability into the concept of ecosystem stability.

As attractive and intuitive as the column _vs._ pyramid metaphor is, it turns out to be wrong. May (1973) showed that, in general, increasing complexity (including diversity) of community trophic structure leads to decreasing stability, a conclusion confirmed by more extensive analysis and computer simulations by Pimm (1982). In fact, increasing the number and strengths of trophic links among species in a community makes the system increasingly less likely to be stable in a mathematic sense. Goodman (1975) and Zaret (1982) suggested that the vaunted instability of the simple ecosystems that Elton (1958) had cited, such as islands and agroecosystems, were a function of other properties of these particular systems, rather than of low diversity per se. Goodman (1975) suggests that the key to whether a particular community is perceived as stable is whether the dominant species (such as the large trees or herbivores) typically undergo enormous vicissitudes of population size, and that whether such explosions and crashes typify such a species is a function of the species' biology, rather than its function as part of a community.

The entire question of community and ecosystem stability is thus no one in which consensus has been achieved. However, it is possible to say that virtually all parties to the debate agree that there is no automatic and simple relationship between species richness and stability. For conservation purposes this finding means that, though maximized species richness might be a goal in its own right (for example, for maximizing potential genetic resources), it is not properly viewed as a goal for its ability to confer stability, even if community or ecosystem stability is a goal.

## MANAGEMENT TECHNIQUES

Just as detailed field studies, and often experimental manipulation, are required to understand the interactions among species, as well as particular species physical requirements, so are such studies necessary to understand what sort of management might be necessary for long-term maintenance of a group of species isolated in a reserve. Such studies, and their results, are necessarily idiosyncratic, but there are many examples in the literature suggesting that management is necessary, and indicating a particular management regime. Fire ecology is a well understood field (see the above discussion of longleaf pine forests), and concomitantly with its growth has arisen a technology of controlled burning and fire management (*e.g.*, Tall Timbers 1976). Many management techniques in addition to fire involve controlled disturbance to maintain disclimax communities (*e.g.*, Pickett and Thompson 1978, Foster 1980).

A growing field is demographic and genetic intervention to counteract the effects of genetic deterioration and demographic stochasiticity in small populations (*e.g.* Goodman 1980, Schonewald-Cox 1983). A limitation of this approach is that, since it is species-based, it is likely to be very expensive, since it will have to be applied separately to each species. Of course both established and experimental technologies of captive propagation (Conway 1980, Benirschke *et al.* 1980), combined with a growing literature on reintroduction (Campbell 1980), may be employed to restore or enhance particular populations in particular reserves. Again it is apparent that these techniques, though they may be effective, will be extremely expensive because of their focus on single species.

Perhaps the only generalization that can be drawn about refuge management is that, in addition to protection, refuges will need management if they are not to change, through either natural succession or progressive degradation of one sort or another. The required management is likely to be much more extensive and onerous for small refuges than for large ones.

Management schemes must always be within the context of a designated conservation goal, and the fact that goals may differ (see above) can lead to conflicting management imperatives. As a general rule, management to maximize populations of particular species may conflict with management to maintain community composition and proportional representation. For example, Szaro (1983) has experimentally demonstrated that conversion of chaparral to habitat more favorable for grazing by large herbivores leads to reduction of biotic diversity. On the other hand, the techniques for realizing single species and community or ecosystem goals may coincide. Harper (1969) found that grazing by rabbits increased herb and shrub species richness on an island community. Management of an ecosystem for increased production is very likely to be at the expense of maximum species richness in the associated communities, since the highest production is often not associated with the most species-rich communities in a sere (Odum 1971). One cannot speak very generally on the question of management technologies associated with different conservation goals because of the idiosyncrasies and complexities of each situation. One can assuredly say that achievement of the community-oriented goal of maintenance of species composition and proportional representation would almost certainly ensure any species-oriented goal and most ecosystem-oriented goals. However, managing a reserve with an eye to single species or ecosystem traits (such as nutrient cycling rates or energy flow rates) is very likely to lead to community changes that would be inconsistent with community conservation goals.

However, management for species or for ecosystems is likely to be technically easier. Even though species' critical interactions may be subtle and numerous, they can be discerned (as noted above) and the appropriate management techniques often flow directly from the study that elucidates the species' ecology. Ecosystem management, though often on a large scale than either species or community conservation, allows wider latitude in exactly what is conserved. For example, if a guild (Root 1967) has been identified as serving a particular function, and maintenance of the function rather than of a particular species has been identified as a goal, it is likely that guild members can substitute ecologically for one another. If short-term utilitarian considerations dictate management policy, there is thus likely to be conflict with individuals whose goal is conservation of communities for their own sake. On the other hand, conservation of communities for their own sake is likely to be the most expensive and technically difficult course.

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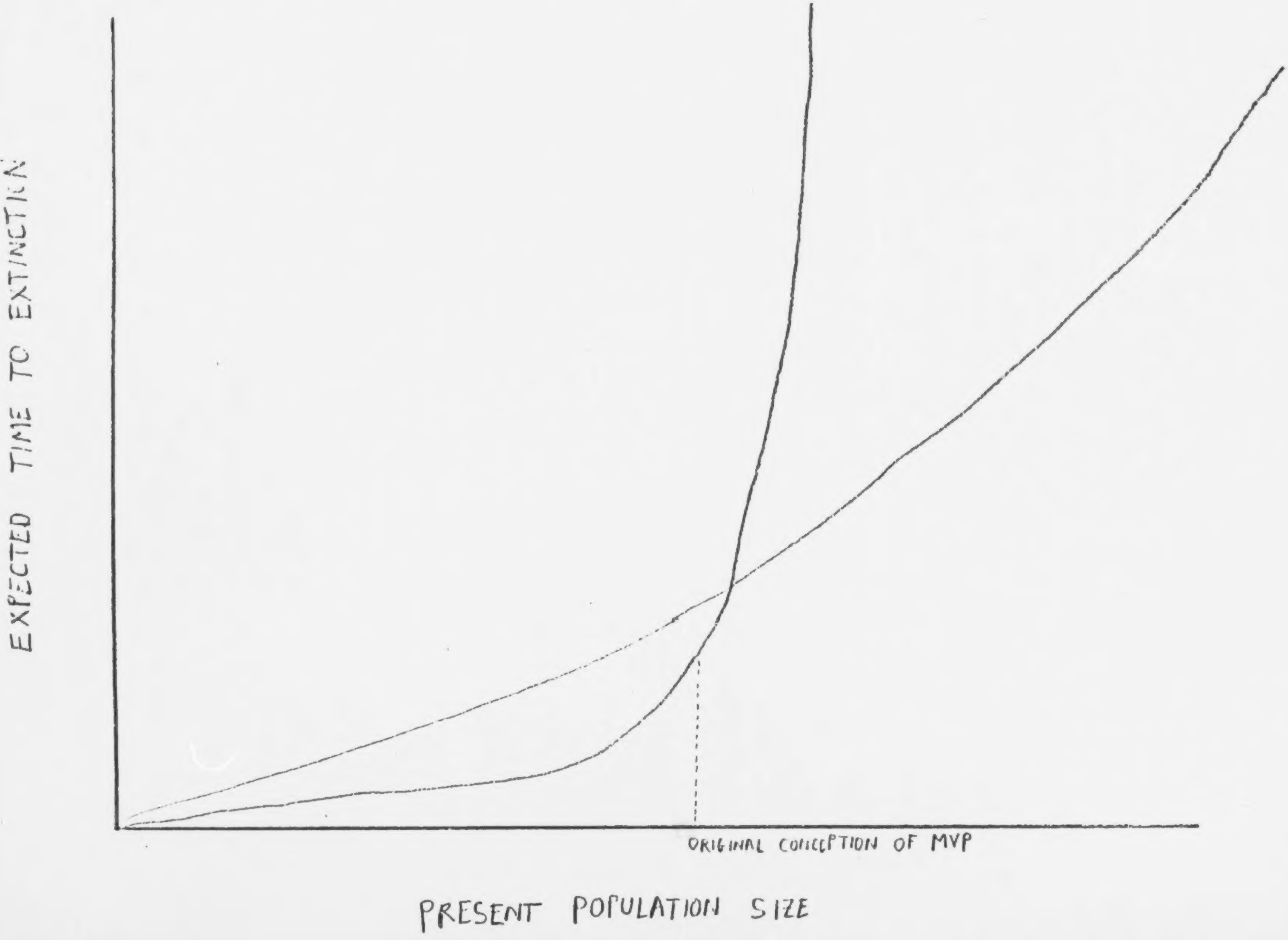

Fig. 1. Relationship of population size to population persistence

Contract Paper No. 5

# Assessment of Application of Species Viability Theories

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Agency for International Development

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# ASSESSMENT OF APPLICATION OF SPECIES VIABILITY THEORIES

A Report Prepared for the Office of Technology Assessment
Congress of the United States

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# ABSTRACT

A viable population is, by definition, one capable of avoiding extinction. Current understanding of the determinants of population viability stem principally from studies of insular biogeography, ecological-evolutionary genetics and theoretical population dynamics. Although no comprehensive, integrated theory of population viability emerges from current knowledge, studies to date have demonstrated that localized extinctions are relatively common events and that the frequency of such extinctions are often related directly or indirectly to population size. Current theory reveals how frequent, localized extinctions are a plausable consequence of such factors as demographic stochasticity, environmental variability and population-wide catastrophes.

Although no single theoretical model of population dynamics incorporates all the above factors, a relative ranking of threats to population persistence is possible. Population-wide catastrophes appear to set the ultimate limit to a population's viability followed, perhaps closely, by environmentally induced year-to-year variations in population growth rate. Demographic stochasticity is only a significant threat to very rare species. The loss of genetic variation due to limited population size is also an important consideration but, based on current understanding, it appears that overcoming environmental and catastrophic pressures may well require population sizes sufficiently large so that the loss of genetic variation is not of immediate concern.

The possibilitites and problems of determining viable population size are illustrated with a case study of the grizzly bear (Ursus arctos). Major management and research needs for assessing population viability and the application of such assessments to the conservation of biological diversity are discussed. Key management needs include development of a consensus on what constitutes successful conservation (e.g., persistence for 10, 100, or 1,000 years, etc.); the development of population viability simulation models, synthesis and review of available data on population size, variability and persistence and systematic compilation of information on the current biogeography of conservation lands. Key research needs include the establishment of long-term population/ecosystem studies, field and laboratory extinction experiments and the enhanced development of stochastic population dynamics theory. The application of population viability theories to ecosystem conservation should focus on determining the area requirements of extinction-prone key species characteristic of the ecosystem in question. Such species will most likely be large-bodied, rare, mammalian or avian predators or those species exceptionally vulnerable to environmental variation and catastrophe.

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# INTRODUCTION

The dictionary defines viability as ". . .the capacity for life, growth and development. . ." As applied to species or their populations, viability is generally understood to mean the capacity for self-perpetuation through time. A species or population is viable if it persists. Thus, by definition, a viable population is one that is capable of avoiding extinction. Until recent decades, an implicit assumption of biology has been that a species' viability was limited only by its ability to adapt, either genotypically or phenotypically, to gradually changing environmental conditions, or by the rare occurrence of paleontologically documented, but biologically inexplicable, catastrophes of a global scale (Raup 1984). Extinction was viewed, with certain dramatic exceptions, as a rare event occurring slowly through geological time in response to some long term, directional environmental change.

In the past several decades, this view of extinction has been expanded dramatically. It is increasingly apparent that, while the above scenario may be true, extinction is also (1) a relatively common event on a local scale, (2) often the results of chance, or at least unpredictable events, and (3) apparently inversely related to population size in its frequency (Diamond 1984).

Recognition of this aspect of extinction stems, in large part, from the equilibrium theory of island biogeography (MacArthur and Wilson 1967) and the wealth of studies the theory has spawned (for reviews see Gilbert 1980, Simberloff 1983). But development in other areas of biological and ecological research have also contributed. In particular, research on the relationship of diversity to stability in ecological systems (Connell and Sousa 1983) and the role of natural disturbance (Sousa 1984) as factors influencing community structure and dynamics have revealed both the distribution and abundance of populations and the structure of the natural communities they form as more dynamic than was previously recognized. Although this emerging picture of the fluidity of populations and communities has neither validated equilibrium theory (Gilbert 1980, Simberloff 1983) nor led to any overally theory of extinction, it has firmly established the study of local extinction and colonization dynamics as fundamental to an understanding of population and community ecology in the broadest sense.

The new view of extinction has also had an immediate impact on conservation thinking (Terborgh 1974, Wilson and Willis 1975, Diamond 1975, Soule and Wilcox 1980, Wilcox 1980, Frankel and Soule 1981, Shaffer 1981). If the likelihood of extinction depends on population size and population size on habitat area, then the quantity of habitat preserved may well determine how long a species survives. With estimates of 15-20% of all species on earth being threatened with extinction by the year 2000 (The Global 2000 Report to the President 1980), due principally to the continued loss of wild habitats, the issue of population size and viability becomes central to conservation efforts. Maintaining viable populations of plant and animal species, in situ, will require a thorough understanding of the relationship of population size, habitat area, and extinction dynamics.

## CURRENT THEORIES ON MAINTAINING VIABLE POPULATIONS OF PLANT AND ANIMAL SPECIES IN SITU

Biological science currently lacks a comprehensive, unified theory of species or population viability and, thus, the extinction process (but see Van Valen 1973, and Stenseth 1979). Current understanding of the extinction process relevant to conservation efforts stems principally from three fields of research: insular ecology (Diamond 1975, 1984, Wilcox 1980, Shaffer 1981), ecological-evolutionary genetics (Soule and Wilcox 1980, Frankel and Soule 1981, Shaffer 1981, Schoenwald-Cox et al. 1983, Lande and Barrowclough in prep.) and theoretical population dynamics (Stenseth 1979, Shaffer 1981, Ginzburg et al. 1982, Goodman in prep., Belovsky in prep., Ewens et al. in prep., Shaffer in prep.). Work in each area has led to certain prescriptions for conservation strategies to enhance population viability, or at least minimize the loss of viability through time (Terborgh 1974, Diamond 1975, Wilson and Willis 1975, Diamond and May 1976, Margules et al. 1982, Franklin 1980, Soule 1980, Senner 1980, Frankel and Soule 1981, Goodman in prep., Belovsky in prep., Shaffer in prep.). A brief review of each area of research and the conservation strategies generated, will place the problem in perspective.

## Insular Ecology

Two general patterns in nature have long fascinated ecologists and biogeographers. First, the larger an area of habitat sampled, the greater the number of species that are generally found in that habitat. Second, islands far from continents tend to have fewer species than islands close to continents, even when both islands are of equivalent area and in similar environmental settings.

The equilibrium theory of island biogeography (MacArthur and Wilson 1967) purported to explain these patterns as the result of an equilibrium between the colonization and extinction of species on islands. Far islands should have fewer species because colonists would be less likely to reach remote areas. That is, distant areas should have lower colonization rates. This is known as the distance effect. Similarly, smaller islands should have fewer species because they would support smaller populations and smaller populations should be more vulnerable to extinction due to nothing more than the probabilistic nature of individual births and deaths (MacArthur and Wilson 1967). That is smaller islands should have higher extinction rates. This is knows as the area effect.

An important corollary to equilibrium theory is that the equilibrium should be dynamic. In the absence of major alterations of island area or isolation, the number of species should remain relatively constant, but the actual species comprising this number should change. This change in species composition over time is termed turnover (MacArthur and Wilson 1967). In addition, equilibrium theory predicts that , if major changes in island area or isolation do occur, a new equilibrium number of species will be attained commensurate with the islands new characteristics. In those cases where island area is diminished or isolation increased the resultant decrease in equilibrium species number is known as relaxation (Diamond 1972).

Equilibrium theory immediately caught the imagination of ecologists and resulted in a wealth of studies attempting to confirm or refute the theory or its postulates (for reviews, see Gilbert 1980, Simberloff 1983). Moreover, the theory was quickly extended from true islands to many terrestrial analogs of islands such as forest patches in agricultural/suburban landscapes (Burgess and Sharpe 1981). In fact, much of the literature on equilibrium theory concerns island analogs, and Wilcox (1980) has suggested, appropriately, that the whole field of study be termed insular ecology to reflect this broader perspective.

The intuitive similarity of nature reserves as islands for those species incapable of surviving in man-dominated habitats was immediately apparent (Terborgh 1974, Diamond 1975, Wilson and Willis 1975, Diamond and May 1976). This intuitive similarity and the findings from some of equilibrium theories' earliest tests led several workers to propose certain strategies for the design of nature reserves that would minimize the loss of species over time (for a review of strategies, see Margules et al. 1982). In general, these strategies stressed the importance of making nature reserves large and close to each other to reduce the impacts of the area and distance effects and, thus, minimize the expected relaxation of reserve biotas over time. Certain strategies concerning the shapes of reserves were also generated (Margules et al. 1982). These strategies are summarized in Table 1.

Table 1. Nature reserve design strategies derived from the equilibrium theory of island biogeography (compiled from Margules et al. 1982). "Better" strategies are thought to maintain more species at equilibrium and/or produce lower extinction rates than "Worse" strategies.

| Better | Worse |
|---|---|
| Large reserve | Small reserve |
| Single large reserve | Several small reserves of equivalent total area |
| Reserves close | Reserves isolated |
| Reserves clustered | Reserves linear |
| Reserves linked by corridors | Reserves unlinked |
| Reserves round | Reserves elliptical |

## Ecological-Evolutionary Genetics

As noted in the introduction, extinction has historically been viewed as a failure of species to accommodate to chaning environmental conditions. Accommodation has both a phenotypic component (adaptation) and a genotypic component (evolution). It is the fundamental theorem of natural selection that the rate of evolution is proportional to the degree of genetic variation on which selection can act (Fisher 1958). Given that any individual carries only a certain percentage of the genetic variation characteristic of its species, it is clear that population size may be a key determinant of maintaining sufficient levels of genetic variation for both the degree and rate of further evolution. The development of gel electrophoresis techniques in the late 1960's uncovered a wealth of genetic variation (protein polymorphisms) that has sparked further speculation on the role of genetic variation in both ecological and evolutionary terms (Nevo 1975).

Small population size can lead to reduced levels of genetic variation in three ways. First, populations may be founded by only a few individuals (e.g., island colonization, population crashes). Because any individual carries only a certain percentage of its species' genetic variation, such populations are immediately constrained to the amount of variation carried by their founders. This is known as the founder effect.

Second, the perpetual limitation of population size to low numbers, as would occur on islands or habitat patches of small size, can lead to the loss of genetic variation simply because there are so few offspring that rare alleles are not represented in some future generation. This chance loss of genetic variation is known as genetic drift. Both immigration of new individuals and strong selection may overcome drift but these forces may not always be present.

Third, continued limitation of population size in the absence of immigration may lead in time to the continued mating of closely related individuals. This is known as inbreeding. Such inbreeding can also diminish genetic variation for obvious reasons and it is well known that close inbreeding in normally outcrossing species (e.g., most vertebrates) leads to both demographic and genetic effects of a deleterious nature (Franklin 1980, Soule 1980, Senner 1980, Frankel and Soule 1981, Ballou and Ralls 1982, Ralls and Ballou 19982a, 1982b).

These considerations have led several workers to approach the issue of population viability on genetic grounds (Franklin 1980, Soule 1980, Senner 1980, Frankel and Soule 1981, Schoenwald-Cox et al. 1983). The basic conclusions from this line of reasoning have been presented by Franklin 1980, Soule 1980, Senner 1980, Frankel and Soule 1981. In brief, inbreeding depression is viewed as the principal threat to short-term survival and the loss of genetic variation from genetic drift as the principal threat to long-term adaptation. Based on the experience of animal and plant breeders, effective population sizes (in the genetic sense, see Franklin 1980) of about 50 are thought necessary to reduce inbreeding depression to acceptable levels. Similarly, based on analysis of genetic variation in quantitative traits, effective populations sizes (again, in the genetic sense) of about 500 are thought necessary to allow natural mutation rates to balance the loss of genetic variation due to genetic drift. Based on this line of reasoning minimum viable population sizes approximating effective population sizes of 50 to 500 are thought necessary to assure short-term survival and continuing adaptation, respectively. Becasue effective population sizes are determined by several factors (sex ratio, variance in individual reproduction, population fluctuations, age-structure - see Franklin 1980, Frankel and Soule 1980, Lande and Barrowclough in prep.), these prescriptions for populations of 50/500 must be translated into actual census population sizes for any particular species. This results in census population sizes larger, sometimes much larger, than 50/500 (on the order of 100s to 1,000s).

## Theoretical Population Dynamics

Biologists have long known that the smaller the population the more susceptible it should be to extinction based on considerations of chance alone. To see this, consider the life of an individual living thing. To fulfill life's potential, the individual must be conceived, go through an intricate developmental period, be born, further develop, mate, reproduce and all the while survive. Anywhere along this continuum of events the individual may die. Death, when it occurs, will always be for some reason (exposure, lack of food, disease, aggression, accident, age) but the reasons are varied and not always intimately connected. Often they involve an element of chance, like falling prey to some predator or being born in the midst of a severe drought. This variety of hazards and their own inherent uncertainty, taken together, make the life of any individual organism a probabilistic phenomenon - a continual roll of the dice. Death is the only certainty, but the time and circumstances are open to wager.

Because populations are nothing more than collections of individuals, extinction is also an inherently probabilistic phenomenon. To be sure, other factors are often involved, but these operate against the ever present background of chance. In simplest terms it is easier to lose two things than three, or three things than four.

Recognition of the role of chance has led to numerous models of stochastic population dynamics designed for, or ammenable to, estimating the time to extinction for a population under various circumstances (for reviews, see Turelli 1977, Shaffer 1978, Ginzburg et al. 1982). In general, these models have focused on the impact of chance events on individual births and deaths (demographic stochasticity: Feller 1939, Bartlett 1960, Bailey 1964, MacArthur and Wilson 1967, Richter-Dyn and Goel 1972) or of chance environmental variability (environmental stochasticity: Lewontin and Cohen 1969, Levins 1970, May 1973, Tuchwell 1974, Turrelli 1977, Ginzburg et al. 1982) or of chance population-wide catastrophes (catastrophic stochasticity: Ewens et al. in prep.).

Stochastic population dynamics presents a complex problem. This is reflected in the fact that, to date, no theoretical models have been developed which simultaneously incorporate the effects of chance at the individual, environmental, population-wide and genetic levels despite the fact that almost every species may experience such effects at each level.

Only recently have stochastic population dynamics models been formulated with population viability applications in mind (Shaffer 1978, 1983, Ginzburg et al. 1982, Wright and Hubbel 1983, Goodman in prep., Ewens et al. in prep.). Most of these efforts have not led to any specific prescriptions for population sizes necessary to assure population survival. Such prescriptions will depend on the characteristics of each species such as density, growth rate, etc. (but, see Shaffer 1978, 1983). However, these modelling efforts have begun to place the role of chance events at each level into perspective in terms of threats to population persistence (Shaffer in prep.). In particular, Goodman (in prep.) has reformulated the classic birth and death process model (Feller 1939) to incorporate both demographic and environmental stochasticity. Ewens et al. (in prep.) have examined some simple models of the effects of population-wide catastrophes on the probabilities of population extinction. Their results are important and bear crucially on the issue of population viability. This can be seen most readily in graphic form. Figure 1 shows a hypothetical plot of the expected, or average, time to extinction ($T_k$) for a population as a function of that population's size (k) for three types of stochasticity: demographic, environmental and catastophic. These relationships are based on the analytical models of Goodman (in prep.) and Ewens et al. (in prep.) as discussed in Shaffer (in prep.). This plot is scaleless because the values of the relationships will vary among species, though the overall form of the relationships are expected to remain the same.

As Figure 1 shows, hypothetical average persistence times increase with increasing population size:

1. geometrically for demographic stochastiticy;
2. linearly for enviromental stochasticity; and
3. logarithmically for catastophic stochasticity.

What this means is that the likelihood of population extinction in relation to population size is least affected by chance events at the individual level and is most affected by chance population-wide catastophes with chance environmental variation being somewhere in between. Put another way, avoiding extinction due to demographic stochasticity may only require a few hundred individuals but larger, perhaps much larger, population sizes will be necessary if the species' environment varies and still larger population sizes may be necessary if the species is susceptible to catastrophes.

An important result of Goodman's and Ewens et al's. work is that there is no universally applicable critical population size that, if achieved, will make extinction virtually impossible. The extinction probabilities of populations and specise will vary and depend most critically on population growth rate, the environmentally induced variability in this rate and the catastrophe scenario to which the population or species is subject. Consequently, neither Goodman nor Ewens et al. give estimates or recommendations of population sizes necessary to assure survival. However, Belovsky (in prep.) has employed Goodman's model along with the allometric (body size) relationships of population growth rate and density and data on the variability of key environmental parameters (e.g., rainfall, etc.) to provide preliminary estimates of populations sizes of mammals necessary to achieve a 95% probability of persistence for two time frames: 100 and 1,000 years. Over a range of 6 orders of magnitude in mammalian body mass ($10-10^6$ grams), population size requirements range from 100's to 1,000,000's of individuals corresponding to reserve areas of 10's to 1,000,000's of square kilometers. As body mass increases, viable population sizes decrease but density relations are such that, the larger the body mass, the larger the reserve area necessary to support the requisite population. The degree of environmental variability can alter viable population sizes for a species of any given body mass by as much as 2 orders of magnitude. Reserve area requirements are, as expected, larger for carnivores than herbivores, and larger for tropical than for temperate species.

It must be stressed that Goodman's, Ewens et al's., and Belovsky's work is preliminary at this point and subject to change. However, even if subsequent work results in reducing Belovsky's estimates of viable population sizes and reserve area requirements by as much as 2 orders of magnitude, demographically and environmentally viable populations of mammals may still reuqire reserve areas in the range of 10's to 10,000's of km$^2$. Very few current nature reserves are 10,000 km$^2$ (Frankel and Soule 1981).

## LIMITATIONS OF CURRENT THEORY

### Insular Ecology

Gilbert (1980) reviewed the literature on equilibrium theory and concluded that few studies actually tested the theory or its major postulates. Of those that did, only arthropod recolonizations of experimentally defaunated mangrove islets (Simberloff and Wilson 1969, 1970) and avian turnover on the California Channel Islands (Jones and Diamond 1976) showed good support for equilibrium theory. Simberloff (1983), focusing on those studies involving birds, came to similar conclusions. Margules et al. (1982) reviewed the literature pertaining to the conservation strategies generated from equilibrium theory and concluded that these are unsubstantiated by evidence or may be shown to lead to equivocal results in practice.

However, it has been clearly demonstrated that (1) for many taxonomic groups in many insular environments, turnover (i.e., localized extinctions and colonizations) are relatively common events that can be documented even in the course of short-term studies (Connell and Sousa 1983, Schoener 1983); (2) in some cases, the rate of such turnover is negatively correlated with habitat area and, by inference, population size (Diamond 1984); and (3) long-term declines in species diversity (i.e., species richness or number) can be documented, or reasonably inferred, for habitats that were reduced in size or increasingly isolated by major habitat fragmentation events of the past, such as post-Pleistocene rises in sea level or the altitudinal retreat of montane vegetation (Brown 1971, Diamond 1984, Patterson 1984). However, it should be noted that such declines occurred in the context of habitats that may have been changing in ways inependent of habitat size or fragmentation.

In the clearest (and only experimental) test of equilibrium theories' applicability to conservation, the World Wildlife Fund's Minimum Critical Size of Ecosystems project (Lovejoy and Oren 1981) has set about to create various size patches of tropical rainforest in the Amazon by coordination with ongoing, local deforestation and development. The project is monitoring the pre- and post fragmentation biotas of patches. Though the project is only several years old and is intended as a long-term (20 yr.) experiment, certain changes in patch biotas are already evident (Lovejoy et al. 1983, Lovejoy et al. 1984). In most cases, this involves the loss of large-bodied vertebrate species. Thus, studies have shown that both turnover and relaxation occur and that their frequencies and/or magnitude are directly or indirectly related to habitat area and, by inference, population size (Diamond 1984). While there are cogent criticisms noting the methodological problems which may artifically inflate (Lynch and Johnson 1974) or deflate (Diamond and May 1977) current estimates of turnover rates, there is little doubt that this is a general phenomenon in nature of potential importance to conservation efforts.

The major limitations of utilizing current estimates of turnover and relaxation rates as guidelines for nature reserve design are threefold. First, not all taxa or environments have been adequately studied. Most turnover and relaxation studies have involved animals, particularly birds, and have focused on temperate and tropical forest habitats. Very little work has been done on plants.

Second, current approaches to assessing turnover are basically site specific, even if this involves numerous sites (Whitcomb et al. 1981). Yet the most meaningful insight to emerge from equilibrium theory is that the occurrence and/or persistence of a species on any particular site may be governed, in part, by regional extinction/colonization dynamics (Levins 1970, Shaffer in press). Most species are probably patchily distributed. Such species can be thought of as a population of populations, or a metapopulation (Levins 1970). The persistence-extinction dynamics of metapopulations have only recently been explored theoretically (Levins 1970, Roff 1974a, 1974b, 1975, Vance 1984). It is clear from this work that both site-specific and metapopulation-wide extinction/colonization dynamics can only be understood in the context of overall metapopulation structure. Most current estimates of turnover rates ignore the metapopulational context in which they are measured (Simberloff 1983) and, thus, loose predictive value if that metapopulation configuration changes.

Third, empirical and experimental measures of turnover and relaxation, to be meaningful, require long term studies relative to the usual sorts of scientific investigations. While Lovejoy et al's. (1983, 1984) experimental approach to studying turnover and relaxation is the most straightforward, this project is intended to last at least 20 years and even then, not all results are likely to be in. Most opportunities to reserve significant quantities of natural habitat will likely be gone within 2-3 decades, so empirical and experimental work involving large scale systems or long-lived species cannot be done in time to bear directly on the problem at hand. This is not to discourage such endeavors; long-term studies of turnover and relaxation will prove crucial to managing whatever remnants of nature society does set aside, but we cannot rely on this approach alone to resolve the issue.

## Ecological-Evolutionary Genetics

Along with attempts to apply equilibrium theory to conservation, genetic considerations have dominated the literature on population viability and conservation (Franklin 1980, Soule 1980, Senner 1980, Frankel and Soule 1981, Ballou and Ralls 1982, Ralls and Ballou 1982a, 1982b, Schoenwald-Cox et al. 1983, Salwasser et al. 1983, Lehmkuhl 1984, LaCava and Hughes 1984, Soule and Simberloff in prep.). This is so for several reasons. First, genetic considerations are of fundamental concern to maintaining population viability, at least in light of the traditional view of extinction and the experience of plant and animal breeders.

Second, the term biological diversity itself has come to imply the full range of life's variety from genomes to biomes. Coupled with the relatively large levels of genetic variation (protein polymorphisms) recently discovered and the emerging technology of genetic engineering, there is a current fascination with genetic diversity, its importance to life, and its potential for benefitting mankind.

Third, and most importantly, the earliest efforts to deal with the issue of population size and viability on genetic grounds led to specific numerical estimates on minimum population sizes for successful conservation (e.g., the $N_e$ = 50/500 rule, see above). Clearly, for the uninitiated planner or manager, such universal, specific prescriptions appear as a God-send.

Unfortunately, this early emphasis on genetics may have been misplaced. As Lande and Barrowclough (in prep.) point out, the population sizes necessary to maintain significant quanitities of genetic variation depend on the type of variation and the form of selection, if any, acting on that variation. Maintenance of typical levels of additive genetic variance in quantitative characters may require effective populaions sizes of only several hundred individuals under directional selection; whereas typical levels of single locus variation which is presently selectively neutral (but may provide the basis for future adaptation to an altered environment) might require effective population sizes of $10^5$-$10^6$ individuals. Clearly, this is quite a range. While there is a general consensus amongst geneticists that additive genetic variance in quantitative characters is the most crucial form of genetic variation for continuing adaptation and evolution, the importance of other forms of genetic variation cannot, as yet, be disregarded.

A further complication with the 50/500 rule is its applicability to other than normally outcrossing species. Many plants and lower animals are self-compatible or tolerant of close inbreeding. In such cases the bulk of a species genetic variation may be distributed between populations rather than between individuals. Consequently, population number may prove as or more important than population size per se. Again, the geographical structure of populations emerges as an important consideration, but this is not explicitly dealt with in derivation of the 50/500 rule.

As noted earlier, application of the 50/500 rule to any real situation will require actual population sizes larger, sometimes much larger, than 50 or 500 ( on the order of 100's-1,000's, perhaps larger: Shaffer in prep.). What is lacking, however, is theoretical, empirical, or experimental evidence to determine the resilience of such theoretically "genetically viable" populations to the effects of the other chance pressures (demographic, environmental, catastrophic) on population viability. Where do genetic population requirements rank relative to the population requirements set by these other factors? The answer is unknown, but as Goodman (in prep.), Ewens et al. (in prep.) and Belovsky (in prep.), have shown there are theoretical reasons for believeing that overcoming enviromental and/or catastrophic stochasticity will require population sizes much larger (perhaps on the order of 1000's to 10,000's for selected species) than those necessary to preserve large measures of additive genetic variance in quantitative traits.

## Theoretical Population Dynamics

To be comprehensively realistic, any model of population extinction in relation to population size should incorporate the effects of demographic, environmental, catastrophic and genetic stochasticity and the impact of geographical population structure, if any, on these effects (Shaffer in prep.). To date, no such model has been developed. The complexity of such a model would quickly outstrip analytical capabilities. Goodman (in prep.) has provided an anlytical model integrating demographic and environmental stochasticity but this ignores catastrophes and genetic considerations and geographic population structure.

Computer simulations employing Monte Carlo techniques may provide a tractable alternative for dealing with such complexity (Shaffer 1981, 1983) but, again, no comprehensive simulation model currently exists. Such models will prove to be very costly. Certain models of the effects of geographic structure on population persistence have been formulated (Roff 1974a, 1974b, 1975, Vance 1984) but these do not incorporate all of the above uncertainty factors.

Aside from the conceptual and logistical difficulties in modelling approaches, the data requisite for their application exists for only a very few animal species and essentially no plant species. For very sophisticated models, data demands may exceed current empirical capabilities (Shaffer 1981, Lande pers. comm.).

Finally, validation of any models developed face the same constraints of time as the empirical and experimental studies of turnover and relaxation necessary for such validation, at least for any but the smallest, most dynamic species. Belovsky's (in prep.) test of Goodman's (in prep.) model against Brown's (1971) data on the extinction of small mammals on Great Basin montane habitat islands is encouraging in the congruence between prediction and observation; but such posteriori tests do not constitute conclusive proof of the models validity, only evidence of its plausability.

Nevertheless, at this juncture, analytical and simulation models coupled with some reasonable interpretation of currently available data, provide the most promising approach to providing initial estimates of viable population sizes and in guiding future empirical and experimental work on population persistence. The following case study provides an example.

## THEORY AND APPLICATION: A CASE STUDY OF THE GRIZZLY BEAR

When Europeans began their exploration and settlement of North America, the grizzly bear ranged almost continuously from the Mississippi River to the Pacific Ocean and from the Mexican Plateau to beyond the Arctic Circle. Today, outside of Alaska, the Yukon and British Columbia, the grizzly remains in only a few semi-isolated populations in the Northern Rocky Mountains (Shaffer 1978). Because of its relations with man, the grizzly is not likely to survive outside those areas devoted to nature conservation (e.g., national forests and other public resource lands). Thus, the grizzly bear is a prime example of a species restricted to conservation lands and whose survival will ultimately hinge on the extent of such lands.

Shaffer (1978, 1983), noting the limitations of standard biogeographic approaches and analytical models of population extinction, developed a stochastic simulation model of grizzly bear population dynamics for use in assessing the relationship between population size and the expected time to extinction. Detailed accounts of the simulation are available elsewhere (Shaffer 1978, 1983). Briefly, the simulation model is a discrete time, discrete number (integer) formulation employing the sex and age structure, mortality and reproduction rates, and density-dependent relationships revealed by an independent analysis of Craighead et al's. (1974) 12 year data based on Yellowstone grizzly bears. Demographic stochasticity is introduced via the discrete number (integer) formulation of the model. Individuals are created and decisions about their survival and reproduction are made through use of pseudo-random number generators. Environmental stochasticity is introduced by use of the variance in the parametric mortality and reproductive rates derived from Craighead et al's. data (Shaffer 1978). Natural catastrophes appear to be unimportant for the grizzly. Lack of adequate information precluded testing the effects of genetic stochasticity.

The simulation could be run in various versions to test the differential effects of demographic and/or environmental stochasticity on population survival. Further, the simulation was used to test the effects of major departures in parameter values or relationships to assess the sensitivity of its results to potential errors in parameter estimation or interpretation.

The results of the basic simulation plus a sensitivity analysis indicates that a population of 50-90 individuals, depending on population characteristics and habitat quality (measured as density), would be necessary to assure a 95% probability of persistence for 100 years. This estimate is most sensistive to an increase in the average mortality rate, increases in the age at first reproduction for females and a shift towards males in the sex ratio at birth. Based on density estimates for the species, reserves of 1,000-13,500 $km^2$ would be necessary to support a population of 50-90 bears.

The original simulation exercise was confined to an arbitrarily chosen 100 year time frame. Subsequent work (Shaffer and Samson 1985) extended the analysis up to 300 years. This work showed that, while a population of 50 grizzlies with the same demographic characteristics as those of the Yellowstone Ecosystem from 1959-71 would have a 95% probability of surviving 100 years, 94% of such populations would go extinct within 300 years. Moreover, the average time to extinction for such simulated populations is only 114 years. These findings demonstrate the crucial importance of the chosen time frame to minimum viable population analyses.

This model represents a fairly crude, brute-force approach to the MVP problem. The simulation is based on only 12 years of data for only one population. Further, genetic considerations have not been incorporated. Nevertheless, it is informative to compare the results of the simulation with predictions from a simple analytical model of population persistence. MacArthur and Wilson (1967) presented a simple model of stochastic population dynamics in their presentation of equilibrium island biogeography. Their model takes into account only demographic stochasticity. That is, the only variation in birth and death rates is that associated with sampling from a binomial distribution. No environmental variation is included. Employing their model for the grizzly bear example cited above provides an estimate of 8336 years as the average time to extinction for a population of 50 grizzlies compared with the estimate of 114 years from the simulation (Shaffer and Samson 1985). This vast difference in expected times to extinction stems from two features of the simulation. First, the simulation model incorporates the sex/age structure of the grizzly bear population. The simple model of MacArthur and Wilson, like most analytical models, does not employ population sex/age structure. Secondly, the simulation incorporates the effects of environmental stochasticity via the variability in mortality and reproductive rates. The simple model of MacArthur and Wilson deals only with average values for these rates. Clearly, more sophisticated models reveal that the likelihood of extinction may be much greater than was previously thought. The addition of genetic considerations would undoubtedly increase this likelihood of extinction, at least for the small populations used in the model.

Above and beyond any application to conservation problems, this finding has major implications for assessing "normal" extinction rates and their bearing on many fundamental problems in ecology and evolution (e.g., biogeography, competition, group selection, speciation, etc.).

## Techniques/Strategies Employed in the Grizzly Bear Analysis

This grizzly bear case study employed Monte Carlo computer simulation techniques to derive an estimate of viable population size for the species. The strategies employed were (1) to recognize extinction as a probabilistic phenomenon and provide an operational definition of successful conservation (e.g., 95% probability of surviving 100 years); (2) analyze available population data on the Yellowstone grizzlies for density-dependent relationships and temporal (i.e., cyclical) patterns; and (3) use the variability in estimated population parameters (i.e., birth and death rates) as a first approximation of the environmental variability to which the species is subject.

The results of the simulation are useful only for the time periods projected and are only as good as the data and analysis on which they are based. Clearly, 12 years of data provides only a limited look at long-term population dynamics. Nevertheless, the simulation is useful as an initial means of screening potential grizzly bear habitat areas to determine which may support marginally small populations and as a means of setting objective recovery targets for the species. It is also useful in ongoing management efforts by providing a means of testing which population parameters are most crucial to determining survival. In this case, the mortality rate is the key parameter. Thus, efforts to lower grizzly bear deaths should provide the greatest increase in expected persistence time for the Yellowstone grizzly bear population.

The grizzly bear simulation was developed primarily as a learning exercise and a demonstration of the potential importance of various factors in determining extinction dynamics. Nevertheless, this work has been employed by the Grizzly Bear Recovery Team (The Grizzly Bear Recovery Plan 1982) to help assess the adequacy of critical habitat area and to set objective recovery targets for this species. In addition, the model and considerations of ecological-evolutionary genetics have prompted the collection of data on levels of genetic variation in this species and modelling attempts incorporating genetic considerations are also being undertaken (Servheen pers. comm.).

Beyond these initial uses, the simulation is available as a continuing management tool for testing the implications of new data on the Yellowstone (Knight and Eberhardt 1985) or other grizzly bear populations and of more refined analysis of the density-dependent relationships of basic population parameters (McCullough 1981). Most importantly, this simulation excercise can serve as a beginning model for other case studies where some detailed population data are available.

## Technical/Scientific Problems

While useful, this grizzly bear case study does not represent a solution either to the viable population problem for the Yellowstone grizzly bears or other grizzly bear populations. First of all, there is evidence that key population parameters may have changed for the Yellowstone population (Knight and Eberhardt 1985). Thus, the analysis should be redone to assess the effects of these changes.

Second, grizzly bear populations are sufficiently different (Shaffer 1978) that the analysis should be repeated using data on the specific population being planned for or managed.

Third, despite the extent of data available on the Yellowstone population, there remain sufficient unknowns so that the simulation is based in large part on assumptions and inference. This is particularly true in terms of projecting future changes in environmental variability. Attempts to apply this approach to other species should recognize that this will remain a fundamental part of the viable population problem.

Fourth, the grizzly bear case study remains simply an unvalidated model and validation is not possible given the nature of the process being dealt with and the impossibility of experimental manipulation and replication.

## Roles of Institutions

The U.S. Fish and Wildlife Service's Endangered Species Program employs recovery teams and recovery plans to manage threatened and endangered species. Recovery teams are composed of representatives of appropriate Federal and State agencies and individuals with relevant expertise on the species in question. Recovery teams develop recovery plans which outline the biology of the species; what is know of the factors causing endangerment; set objective levels for distribution and abundance of the species which, if achieved, would constitute recovery from endangerment; and outline the tasks to be accomplished to achieve such recovery. Institutional roles and responsibilities for accomplishing these tasks are also identified in recovery plans.

Numerous agencies at the Federal, State and Provincial level are involved in the grizzly bear recovery effort besides the U.S. Fish and Wildlife Service. These include the U.S. National Park Service, the U.S. Forest Service, the U.S. Bureau of Land Management, The States of Wyoming, Idaho, Montana and Washington, and the Wildlife Branch of the provinces of British Columbia and Alberta. In addition, there are representatives from the U.S. Bureau of Indian Affairs and the Salish Kootenai and Blackfeet Tribes. These agencies are organized to implement the grizzly bear recovery plan, which involves coordinating all research and management necessary to recover the species. This organization is accomplished through the Interagency Grizzly Bear Committee composed of high-level officials from each agency. This committee integrates all management and research to accomplish the recovery of the grizzly bear.

It is important to note that the viable population analysis just discussed did not evolve from these interagency efforts, though the analaysis was used in the Recovery Plan. To date, no formalized protocol for dealing with the viable population problem has been adopted by the Service's Endangered Species Program. However, the U.S. Forest Service is moving in this direction (Salwasser et al. 1983) and, eventually, all conservation efforts involving considerations of population size and habitat area will need to address this issue. The key need in this area is not so much in fostering greater cooperation, cooperation already exists (however, this may not be true in other parts of the world). The crucial need is in developing some biologically and socio-economically acceptable consensus on the level of security sought for endangered species and the time frame to which this security is to apply. Once these issues are addressed, viable population analyses, as exemplified by the grizzly bear case study, can aid in better determining what management needs exist.

## Application to Grizzly Bear Management

As noted earlier (p. 20) minimum viable population (MVP) theory's principal application to grizzly bear management in the Northern Rocky Mountains has been in helping establish target recovery levels for the species rather than in any application of field management programs. However, it should be noted that, because Section 7 of the Endangered Species Act requires a finding of no jeopardy to a Threatened or Endangered species before Federally financed activities can take place in areas designated as critical habitat, such target recovery levels have far-reaching, if indirect, effects on numerous resource management programs. Such diverse activities as timber harvest, oil and mineral exploration or back-country recreation must take into account the effects of such activities on maintaining or reaching recovery targets. Consequently, such direct management practices as habitat conservation or enhancement, road-building, depredation control, or harvest may be influenced by MVP analysis (Servheen pers. comm.).

More important is the role MVP analysis will play in the future management of species confined to fragments of their former range. Management interventions for each species are likely to be proportional to population size. The smaller the population, the more management activity will be required to diminish the probability of chance extinction. For example, inbreeding may have to be countered by occasional transfer of individuals from one isolated population to another. Either death rates or birth rates or their variability may need to be influenced to overcome the pressures of environmental stochasticity. Or, if all else fails and the species is lost in one isolated population, perhaps due to localized catastrophes, reintroduction efforts will be necessary. MVP analyses, as they become more sophisticated, can aid significantly in all these areas. Such analyses can reveal which preserves and population are most secure, when reintroductions or supplemental releases may prove most effective and the size of reintroductions likely to be successful (see Goodman in prep.). For example, the grizzly bear MVP analysis indicated that extinction probabilities were most affected by small changes in the population mortality rate. Thus, efforts to control grizzly bear mortalities should prove most effective in enhancing the Yellowstone population's chances for survival. Knight and Eberhardt (1985) have reached similar, though more refined, conclusions based on the long-term Interagency Grizzly Bear Study Team's research program. They note that adult female survivorship is likely the key factor in this population's survival.

## Application to other Situations

The grizzly bear MVP analysis has been useful primarily in elucidating the problem of species conservation where habitat area and population size are key issues; providing a conceptual framework for defining population viability; and demonstrating what can be done with available information to better assess the possible long-term consequences of short-term management decisions. As noted earlier, this analysis is not a final solution, but it is a useful conceptual framework and management tool.

Groves and Clark (in prep.) have reviewed various approaches to MVP analysis for the black-footed ferret (*Mustela nigripes*). Applying the suggested approach of Lehmkuhl (1984), they concluded that a population of 200 black-footed ferrets would be the minimum size population considered viable. This figure of 200 for the black-footed ferret represents adjustments in the census population (for variance in progeny number, unequal sex ratios, overlapping generations and population fluctuations) to yield an effective popualtion size ($N_e$) of 50. Thus, Lehmkuhl's approach is based on the genetic viewpoint that inbreeding depression is the principle threat to population viability. As previously discussed, this may prove incorrect.

Salwasser et al. (1983) have used a similar genetic approach for determining target population management levels for the northern spotted owl (Strix occidentalis caurina) but with the addition of considerations of population dispersion characteristics. Their conclusion was that 500 pairs of breeding spotted owls should assure perpetuation of the species in the Pacific Northwest. However, this effort also seems to implicitly assume that genetic drift is the principal threat to population viability.

Berry (pers. comm.) has assessed the population viability density requirements of the desert tortoise (Gopherus agassizii) in California. Berry believes desert tortoise populations require at least 100 individuals per square mile or 50 adults per square mile with a total population size ranging from several hundred to several thousand individuals to assure long-term viability. The necessary habitat for such a population should be in one contiguous area and not fragmented. The habitat should also be relatively undisturbed. Disturbance or habitat fragmentation would require increases in aggregate population size to assure viability.

Berry's recommendations are based, in part, on genetic considerations but not on a quantitative model of the species' population dynamics. Walter (pers. comm.) has developed a quantitative model for assessing the survival requirements of the desert tortoise, but this is still in the review process and is not yet available from the author.

What is lacking in all of these efforts is a quantitative model of population dynamics that can aid in assessing whether the inherent genetic assumptions will in fact prove more crucial then demographic, environmental or catastrophic considerations. Perhaps Walter's work addresses this need.

The major principles that have emerged from these MVP analyses that are applicable to other management situations are (1) the need for an operational definition of what constitutes successful conservation, (2) the need to recognize the various types of inherent natural variation as important determinants of expected population lifetimes, and (3) the need to develop more comprehensive, sophisticated models of population dynamics applicable to a wide range of species with only limited data availability. More case studies will be useful, but a concerted effort at developing generic models of population persistence in relation to population size, habitat area, geographic structure and the types of natural variability (demographic, environmental, catastrophic and genetic) are urgently needed if intelligent decisions on habitat conservation are to be made in time to be effective.

## HOW CURRENT KNOWLEDGE CAN BE BETTER UTILIZED AND IMPROVED

It should be clear from the preceding sections that ecological science is undergoing something of a revolution in understanding extinction dynamics, both local and global. This revolution is occurring both at the empirical, experimental and theoretical levels and, as just discussed, is being integrated into current efforts at conserving and managing biolgoical diversity. It should also be clear that the revolution is still in progress and much remains to be learned. Several approaches may be used to better apply current knowledge concomittant with enhanced efforts to refine and expand our understanding of extinction dynamics. The imminence and potential magnitude of the extinction crisis mandates such a dualistic approach.

### Better Utilization of Current Knowledge

Much knowledge exists that can aid in conservation planning but enhanced efforts are necessary for its effective utilization. Efforts should concentrate on four areas: establishing conservation criteria; developing management models; synthesis and review of available population data and population dynamics relationships; and systematic compilation of information on the current conservation biogeography. Each is discussed in turn.

Conservation Criteria. . .The dependence of population persistence on population size and time immediately begs the question of what constitutes successful conservation. Is a 95% chance of surviving 100 years sufficiently secure or woefully vulnerable? Many conservation land-use decisions will hinge on the answer to this issue.

Biology, particularly palebiology, and evolutionary ecology may provide guidance, yet the issue cannot be answered solely on biological or scientific grounds. The issue is sufficiently crucial to conservation efforts that thought should be given to convening a committee of experts, under the aegis of the National Academy of Science (NAS), but with participation from such other fields as sociology, economics, political science, medicine and law. The Committee should fully explore and discuss the options open to society and their implications. A report of the Committee's deliberations, opinions, and recommendations should be published.

Management Models. . .By definition each species is unique. Yet, taxonomy itself is proof of the heirarchical similarities in key characteristics among species. Just as with anatomical features, species may be classified according to life-history type (e.g., discrete vs. overlapping generations, continuous vs. periodic reproduction, density-independent vs. density-dependent birth and death rates, etc.). Given current biological knowledge and computer capabilities, generic simulation models of population persistence in relation to demographic, environmental, catastrophic and genetic stochasticity could be developed for major life-history types. Such models should be interactive, require a minimum of data and provide maximum flexibility in assessing the effects of various data and population dynamics relationships. Models should be designed for use on widespread mini and micro computers and be readily understood by users with a minimum of computer experience. Availability of such models would allow any agency, organization, institution or individual to rapidly and effectively organize the available data on the species or population being considered, determine if sufficient data exists to make a preliminary viable population assessment, identify key gaps in data or relationships, and provide a sensitivity analyses of which data gaps are most important. Such models would provide a much needed "what-if" capability to conservation planners and managers for exploring various management options. Moreover, availability of a series of standardized generic models would facilitate comparisons among different situations. Some efforts in this direction have already been made (Samson pers. comm., Harris 1984) but an expanded approach is needed.

Such models should not be viewed as a solution to the viable population problem but as a much needed management tool to deal with the complexity of population viability. Availability of such models would spur more case studies similar to the grizzly bear example discussed previously.

Such an effort would likely be most effective if done by a leading software company working under specifications developed by a panel of ecologists, wildlife managers and population dynamics experts. Such a panel could be appointed by the NAS.

## Synthesis and Review of Population Data and Relationships

Much information exists in the literature on basic population data (density, reproduction, mortality, etc.). As Belovsky (in prep.) notes, such basic population characteristics often bear a predictable relationship with basic species' characteristics such as body size, at least for mammals. Allometric relationships of this sort can prove useful in providing first approximations of the population dynamics characteristic of species for which there is no data. Also, for many species, there may be direct measures of some population characteristics but not others (e.g., reproduction vs. mortality in birds). Because of the burgeoning number of cases to be dealt with and the difficutly, expense and time required to gather detailed population data, concerted efforts should be made to compile, review and interpret available population data for major groups (e.g., mammals, birds, reptiles, amphibians, fishes, etc.). Work dealing with such aspects of population dynamics as density-dependent versus density-independent population regulation, inter-specific competition, levels of environmentally induced variation, and the frequency and extent of population-wide catastrophes, etc., should be reviewed and summarized to indicate what relationships have been conclusively demonstrated; what patterns there may be in such relationships for species of differing life-history types or in different ecological situations and, absent any conclusive results, the general consensus of experts on the likely form of such relationships and patterns. This compilation, review and interpretation work is crucial to the successful development of generic MVP management models and should precede or be concommitant with that effort. Again, the NAS could provide the institutional framework for accomplishing this task.

Current Conservation Biogeography. . .Conserving elements of biological diversity threatened by habitat loss obviously entails consideration of how much habitat to conserve, how distributed and how inter-related. As the discussion of viable population estimates for mammals revealed, some species may require very large population sizes and/or reserve areas to assure a high probability of long-term persistence. In some cases, these requirements may exceed political, economic or social capacities. In such cases, the most feasible management alternative is to assure the representation of the species in more, perhaps many more, than one reserve, any of which might prove too small to assure independent persistence. Such redundancy, couple with intensive management, may be the only means of assuring the survival of some species. Consequently, as national and international efforts proceed to reserve additional lands for conservation, the current distribution and abundance of target species on current conservation units becomes a crucial consideration in prioritizing future designations. Many conservation units do have historical and/or current distribution and abundance data. However, a much expanded effort is warranted to describe the current conservation biogeography as a means of its improvement. The IUCN's Conservation Monitoring Center is working in this direction and might be an appropirate institution to undertake an expanded effort. The Nature Conservancy's Natural Heritage Program and the International Program could serve as conceptual models for such efforts.

## Improving Current Knowledge

Despite best efforts to fully utilize current knowledge, we simply know too little to have much confidence in current determinations of viable population sizes or their area requirements. While efforts must proceed to deal as effectively as possible with the viable population issue based on current knowledge and theory, serious efforts at improving our knowledge and theory are equally crucial to conservation in the long term. Though opportunities to create major conservation units or systems may be foreclosed within 2-3 decades, management of whatever units are reserved will be an ongoing challenge. If application of current knowledge and theory is seriously in error, only an enhanced understanding of extinction dynamics will help rectify the resulting problems.

There are three principal approaches to enhancing current knowledge of extinction dynamics: (1) long-term population/ecosystem studies, (2) field and laboratory habitat fragementation/extinction experiments, (3) development and refinement of stochastic population dynamics theory. Each approach should be pursued aggressively.

Long-term Population/Ecosystem Studies. . .Scientific experiments and management applications are often conducted in a limited temporal framework. For example, studies may be conducted to determine what factors limit the abundance of a species in a particular area over a certain study period. Based on the findings of such studies, management may alter relevant factors to enhance or restrict the target species abundance in accord with managment's objectives. But this entails an implicit assumption that today adequately represents history and/or the future. What regulates a species' distribution and abundance may well change from place to place or over time. In fact, evolutionary theory itself implies such geographic and temporal change. Consequently, long-term conservation demands long-term studies of the factors affecting the persistence of populations.

Ongoing long-term studies can significantly improve understanding and reveal factors and relationships undetected by shorter studies. Good examples are the work on Checkerspot butterfly populations (Ehrlich and numerous co-workers - see Ehrlich 1983), the Isle Royale moose-wolf predator-prey system and the Hubbard Brook Experimental Forest ecosystem studies (Likens 1983). NSF's Long-term Ecological Research program (Callahan 1984) is a recongition of, and an attempt to deal with the need for more long-term studies.

Such efforts should be extended world-wide and enhanced to further incorporate studies of those factors impacting population lifetimes. Especially important to such an effort is the establishment of a spectrum of studies to adequately represent a range of taxonomic/ecological levels and situations and a meaningful range of population sizes. UNESCO's Man and the Biosphere Program and the world-wide system of Biosphere Reserves would be an appropriate focus for such efforts.

Field and Laboratory Extinction Experiments. . .Ideally, scientific determinations of population lifetimes would entail experimental manipulations of a large number of replicate populations spanning several orders of magnitude (say 6, $10-10^6$) in size for a period sufficiently longer than the conservation objective (say 1,000 years if the objective is 100) to ensure that a representative span of temporal conditions and/or cyclical population behavior may be captured in the results. Realities render such an approach fantasy but experiments, on an appropriate scale, may still make a meaningful contribution. The World Wildlife Fund's Critical Size of Ecosystem Project (Lovejoy and Oren 1981) mentioned earlier is the clearest example. By combining pre-disturbance inventory work with post-disturbance monitoring of a wide range of forest remnants, the project is already documenting short-term biological and physico-chemical changes that occur as tropical forests are fragemented by development. Despite the fact that many long-term consequences remain to be documented, this approach is already providing useful results and is likely to continue to do so. This approach warrants duplication in other ecosystems, interwoven with ongoing development or management such as timber rotation schedules on multiple use resource lands.

A second potentially profitable approach is lab experimentation with very short-lived, highly dynamic species such as insects (for example, see Philippi et al. in prep.). Goodman's (in prep.) model of extinction probabilities, incorporating demographic and environmental stochasticity, reveals such sensitivity to the latter, particularly for certain ranges of growth rate and its variability, that experimental tests of the model's projections should be possible in the short-term. While confirmation of predictions from the model over only a limited range of situations (e.g., small populations with low growth rates subjected to extreme levels of environmental variability) would not constitute direct validation of the model over untested ranges of parameter values, experimental work showing a positive relationship at some scale would certainly enhance the attactiveness of the model for providing first approximations of viable population sizes for those situations which are not empirically or experimentally testable.

Development of Stochastic Population Dynamics Theory. . .As repeatedly pointed out, empirical and experimental work on population lifetimes are necessary but not sufficient components of dealing with the viable population problem. Conservationists will continue to rely on theory even as this advances through empirical and experimental work. The key theoretical need for addressing the viable population problem is in development of comprehensive, realistic models of stochastic population dynamics capable of simultaneously integrating the effects of individual, environmental, catastrophic and genetic uncertainty, both for single isolated populations and for metapopulation configurations. For single populations, such models are currently feasible via computer simulation techniques. For metapopulations, even computer approaches are in their infancy. However, provided sufficient interest, direction and resources are forthcoming from the institutional sources of research support, progress in this area can be expected to be more rapid and yield more immediate and substantial dividends than in any other area.

One area within theoretical development should particularly be stressed. The initial response to a call for more sophisticated theoretical models will be such models requiring very detailed population data that may well outstrip our current abilities of measurement in the field. Some effort should be made to assess the limits of our empirical abilities, in terms of the detail and resolution of population data, which are likely to continue. Theoretical efforts could then focus on the most appropriate ways of dealing with uncertainty in the context of realistically available data. In other words, rather than endlessly pursuing greater accuracy and precision, can we learn to manage effectively with realistically available data?

## APPLICATIONS TO ECOSYSTEM CONSERVATION

The focus of this paper has been on the determinants of the survival of species and their populations. Emphasis has been on the role of population size and number and, by inference, habitat area as key determinants of population lifetimes. No discussion has been included on habitat quality or community-ecosystem structure. Nor has the broader issue of community or ecosystem conservation been explored. Recall that estimates of impending extinctions range from 500,000 to 2,000,000 species within the next several decades. Most of these extinctions are thought likely due to continued habitat loss. Species by species approaches to determining population size and number and habitat area to assure successful conservation would be an exhaustive, impractical approach based solely on the volume of cases to be examined. Further, many of the species likely to go extinct have not been described taxonomically and the details of their life-histories remain unknown. Taken together with the intricate inter-dependencies of many species, these factors argue for a community or ecosystem approach to the conservation of biological diversity. Yet such approaches are plagued with their own difficulties not least of which are attempts to classify communities and ecosystems. While most ecologists no longer view communities or ecosystems as discrete units of organization (i.e., analogous to the highly integrated individual), there are repeatable patterns of structure, function, and, in some cases, species co-occurrences, that produce patterns in the living world. For example, plant species composing the major vegetational elements of some communities have been shown to be distributed independently of each other by species-specific responses to complexes of environmental gradients (Whittaker 1970). Yet, many parasites, parasitoids and plant-pollinator complexes are highly co-evolved species' assemblages in which the distribution and abundance of one species depends on that of another. Focusing on plant conservation may lead one to believe that the broad-scale patterns manifest in nature are, too a large extent, probabilistic and describable only in statistical terms. Consideration of co-evolved species complexes may lead to the opposite view.

How then can community/ecosystem conservation be approached and how can viable population strategies be utilized in the effort? Systematic conservation will require three broad efforts:

1. A classification of the elements of diversity (species, communities, ecosystems, and biomes).
2. Assessment of the size, number and location of reserves necessary to conserve targeted elements of diversity with some agreed upon level of security.
3. Active management programs for reserves to perpetuate their natural dynamics that may be disrupted due to constrained reserve size or extensive or intrusive non-reserve habitat alteration.

The first effort is a problem in taxonomy, community ecology and biogeography and will not be dealt with here except to note that there is sufficient current knowledge so that a useful, integrated classification system of terrestrial biological diversity should not present a major or lengthy problem provided conservationists recognize and accept two facts. First, with 500,000 to 2,000,000 species potentially in jeopardy, preventing all extinctions is not possible. Reducing or minimizing extinction is the only feasible goal and some difficult trade-offs will be necessary. Second, no classification system will sufficiently capture the nuances of nature to universally satisfy all interest groups or experts. Nevertheless, the major patterns of nature can be adequately described by current classifications for many, if not most, areas. The classification system employed by the Biosphere Reserve system is a practical case in point (Gregg 1984).

Assessments of the size, number and location of reserves and development of management strategies and programs for reserves will both benefit from population viability considerations. Based on current understanding, we know that the rarest (i.e., lowest density) species and/or those most subject to environmental variability and/or catastrophes are the species which will require the largest population sizes and/or largest reserve areas to assure a high probability of long-term persistence.

Attempts to assess the area requirements of communities and ecosystems should focus on those species representative of the community or ecosystem which are most at risk to chance extinctions. In most cases, these will be large-bodied carnivores or omnivores or those species with pronounced, perhaps cyclic, population fluctuations. If we are successful in providing sufficient room for such species to survive, then the other, less-space demanding, members of their communities may also survive. For example, East (1981) has summarized the available data on the density and distribution of major mammalian herbivores and carnivores in African savanna reserves. Based on species density and species area considerations, it appears that reserves large enough to maintain representative populations of such rare predators as wild dog (Lycaon pictus) or striped hyenas (Hyaena hyaena) would also capture representative populations of the vast majority of native herbivores. However, the viability of such representative populations is problematic and in need of analysis similar to the grizzly bear case study.

If the generic models of population persistence discussed in the preceding section were available, they could be used to approximate MVP sizes for key area-sensitive species. Based on density data, these estimates could be translated into area requirements for the key species and an initial assessment made of the population sizes of other species provided for by such an area. Such an approach is not likely to adequately provide for all natural diverstiy but it may provide for the bulk of diversity and may be the only meaningful alternative in light of the magnitude and complexity of the problem. Certainly, several case studies could be done to test this approach.

In sum, a general classification of biotic diversity at the community/ecosystem level could be readily developed. For each element in such a classification, key species representative of that element, yet prone to extinction, could be identified. Case studies of such species could produce an estimate of viable population size and, in some cases, number. These area estimates could be used in conjunction with available information on species distributions and densities to determine what other community members would be contained in such areas and in what numbers. For those species not adequately represented by such a system additional reserves would be necessary to fill in the gaps. Such additional reserves could focus on narrowly endemic species and unique or rare community types. Slowly, but steadily, such an approach could flesh out a representative and, to a certain point, enduring nature reserve system to aid in maintaining biological diversity *in situ*.

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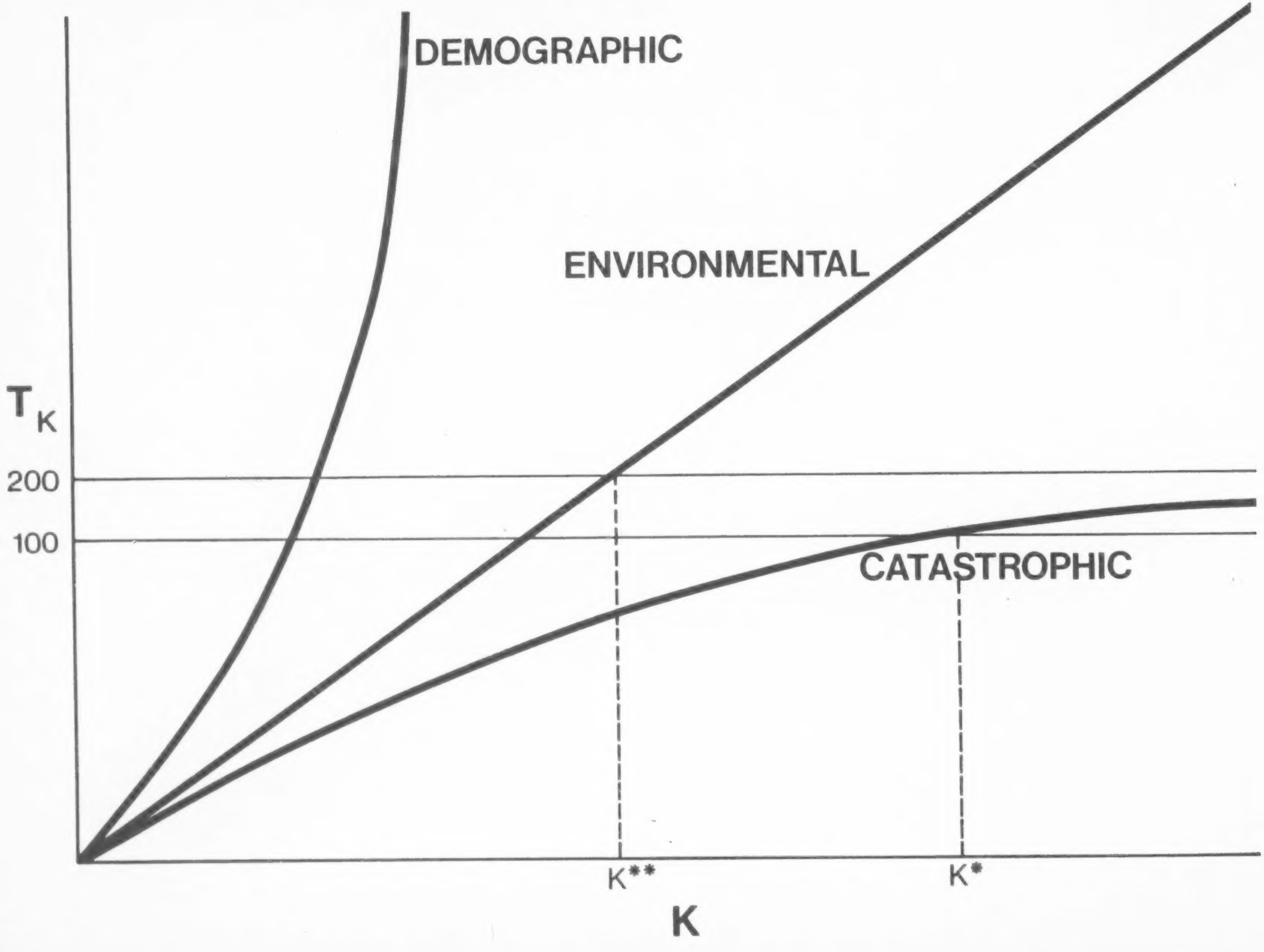
Figure 1. An idealized portrayal of the functional form of the relationship of the expected (average) time to extinction of a population ($T_k$) as a function of the population's carrying capacity (K) for three types of uncertainty: demographic, environmental, and catastrophic. For a 100 year period, a population K* would be large enough so that the average population would survive the effects of all three types of uncertainty. For a 200 year period, a population K** would be large enough so that the average population would survive the effects of demographic and environmental uncertainty. However, no population would survive catastrophic events. In such a case, multiple populations are the only management alternative. Note that these functions are based on the average time to extinction. However, the time to extinction under the combined effects of demographic and environmental uncertainty is exponentially distributed. Thus, to assure a high probability of a particular population surviving, that population size would have to be increased (adopted from Shaffer in prep., based on the models of Goodman in prep., and Ewens et al. in prep.).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Contract Paper No. 6

# Diversity of Marine/Coastal Ecosystems

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# PRESERVING THE DIVERSITY OF MARINE AND COASTAL ECOSYSTEMS

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# ABSTRACT

Diversity in marine and coastal ecosystems includes the concepts of both ecosystem and genetic diversity. Ecosystem diversity must incorporate both functional and taxonomic components and involve consideration of life forms (eggs, larvae, juveniles, adults) and habitats.

Strategies for protection of biological diversity in marine and coastal ecosystems have not been specifically articulated. The theoretical framework for development of these strategies primarily revolves around identification and preservation of representative habitats within coastal and maritime provinces.

No specific mandates to maintain biological diversity in marine and coastal ecosystems have been developed by the United States. None of the international arrangements for protection or conservation of biological diversity to which the United States is a party deal exclusively with marine or coastal ecosystems. A number of Federal programs do, however, specifically address protection and preservation of coastal and estuarine habitats (Estuarine Sanctuary Program, Marine Sanctuary Program, National Park Service Programs). Our ability to fully preserve or protect biological diversity in marine and coastal ecosystems is hampered by our inability to accurately and economically conduct resource assessments in these areas. Although resource assessment techniques have developed rapidly in recent years, resource assessment in coastal and marine regions is predominately descriptive. The full potential of image processing, remote sensing, computer science and modern information analysis and presentation technology has not been realized, primarily because of limited economic resources to apply to the task.

Traditional practices for protecting genetic diversity in terrestrial plant and animal species such as storage and banking of germ plasm, and establishment and maintenance of botanical gardens and zoos have not been applied to protection of genetic diversity of marine and coastal species.

Recommendations for improving the position of the United States with regard to preservation of diversity in marine and estuarine ecosystems include: requiring that terrestrial oriented habitat protection programs recognize and address abutting marine and estuarine habitats; requiring management plans for federally protected marine and estuarine areas address protection of biological diversity; establishment of an interagency task force to provide a periodic combined assessment of protected marine areas; and encouragement of improved resource assessment technology. The most important factor in the failure to develop a comprehensive program to protect marine and estuarine diversity is identified as lack of concern for the problem. An adequate public awareness program is recommended to overcome this concern.

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# DIVERSITY OF MARINE/COASTAL ECOSYSTEMS

## Introduction

The term "diversity" as it relates to marine/coastal ecosystems can cover at least two separate but related concepts. Genetic diversity is an estimate of the amount of genetic variability among individuals in a single species and between species. Ecological diversity is an estimate of the number of species in a community, their changing abundance and the various processes and interactions within and between communities. The term biological diversity, in its broadest sense, includes both genetic and ecological diversity (WRI, 1984). In addition, Ray (1985 -- after Hutchinson, 1959) argues that diversity is functional as well as taxonomic and involves both life forms and habitats. The concept of including life forms as opposed to simply species has particular importance in the consideration of diversity in the marine environment.

For purposes of this paper, we will focus primarily upon ecological diversity and how consideration at this level is inseparable from a consideration of the processes interacting in marine/coastal regions and appropriate for maintaining marine and coastal species diversity. Alterations of ecological processes and destruction or modifications of habitats at sea, but more particularly in coastal areas, constitute the major overall threat to diversity at all levels. Direct endangerment to species (with the exception of marine mammals and marine turtles) is not nearly as much of a concern in coastal and marine systems as in terrestrial systems.

## Land versus Sea

Differences in diversity. Land contains approximately 4 times the number of species found in the sea. Consideration of species numbers alone would argue that lands in terrestrial ecosystems are inherently more diverse. Differences in faunal diversity between marine and terrestrial environments are, however, primarily due to insects. Hutchinson (1959) indicates that if insects were removed from the picture, marine fauna would appear more diverse than terrestrial. Terrestrial plant species clearly provide greater diversity than marine flora.

If, however, one were to view diversity from a different perspective, that of number of higher taxa (particularly animal), one could view the sea as more diverse. Many higher taxa (phyla, classes, orders) are exclusively marine. Implicit in this view that the number of higher taxa represents true diversity is the consideration that the higher the level, the greater the genetic difference, i.e. a single species may be the sole representative of an order, class, or phylum, and that the loss of one of these species may represent a far greater genetic loss than a species in a taxon made up of several hundred or thousand members. (Consider, for example, the loss to society if the single representative of the Order Xiphosura, the horseshoe crab, Limulus polyphemus had become extinct before its use to the medical profession were realized.) From a genetic point of view, a species level examination of diversity in the marine environment can be highly misleading (see Ray, 1985).

An additional factor of importance when considering techniques or methods of protecting biological diversity, is that many of the fish and invertebrates that make up the bulk of marine species pass through several life stages from egg to adult. Many of these life stages act as if they were different species than the adult form. These "different" forms may inhabit a different ecosystem or distinctly different niche within the same ecosystem. For example, in the Chesapeake Bay, the adult Blue crab, _Callinectes sapidus_ occupies the entire estuarine part of the Bay, even extending occasionally into fresh water. The adult females spawn in the higher salinities of the Bay mouth and the hatched zoea migrate to the upper water layers, and are carried offshore as part of the inner shelf neuston. The last larval stage, megalops, are found primarily along the bottom, particularly in beds of submerged aquatic vegetation in the lower Bay. The extensive migrations of anadromous species such as salmon, striped bass and herring which spawn in fresh water frequently far inland from any influence of the ocean and mature in mid ocean, and the catadromous eel which spawns in the region of the Sargasso Sea and, depending on the species, matures in the fresh waters of North America or Europe also point to the difficulties which must be considered in trying to protect biological diversity in marine and coastal regions. In order to maintain any one species, all of its life stages must be maintained. The problem of diverse life history stages is also a problem with the terrestrial insects, but is rare among many of the land species which dominate our thinking.

_Ecological differences._ What are the principle differences between terrestrial ecosystems and marine and coastal ecosystems beyond the obvious presence of more water (usually salty) in the latter? Pielou (1979) observed that: (1) vegetation does not form a structured environment in most marine systems; (2) the oceans are contiguous and marine species have greater ranges, generally than terrestrial species; (3) the boundaries to dispersal are more subtle, but the closer to land, the clearer the boundaries may become; (4) latitudinal zonation is more marked for marine than for terrestrial species; and (5) the ocean is three dimensional, with biogeographical zonation a matter of depth as well as geographical location.

The overriding difference between terrestrial and aquatic environments is, of course, the difference in the relative proportions of solids, liquids and gases. Marine ecosystems are dominated by liquid (sea water) while coastal environments encompass both terrestrial and aquatic attributes (marshes, dunes, intertidal flats, shallow water, deepwater, oyster reefs, coral reefs, hard bottoms, soft bottoms, kelp beds, submerged vegetation beds, mangroves, maritime forests etc.). The characteristic dynamic and kinetic properties of the water molecule result in the fact that chemicals, organisms, nutrients, sediments, entire communities, and even ecosystems can be suspended, transported, and mixed in three dimensions over varying periods of time and space that are fundamentally different from those of the largely in-air world of terrestrial biota.

Only benthic marine communities tend superficially to act somewhat like terrestrial environments. The biology, ecology or physiology of marine organisms is frequently drastically different from terrestrial organisms. As an example, on land, the primary producers, plants, are usually the larger more visibly dominant species, whereas in the sea, the larger, more visibly dominant organisms are usually animals. Further, turnover rates and energetics are markedly different; on land, the greatest accumulated biomass and longevity occurs among plants, whereas in the sea, animals bear these characteristics.

The physical differences between land and sea also dictate differences in physiological requirements. On land life processes are often dominated by temperature fluctuations because the thermal capacity of air is small. In the sea, thermal capacity is great and diurnal temperature fluctuations are not as great except near the surface. In addition, seasonal and latitudinal temperature differences are not extreme. Also, the oxygen capacity of air is much greater than in water. Aquatic organisms need to process about 20,000 ml of water to obtain about 100 ml of oxygen while this amount of oxygen is available in only 500 ml of air.

Terrestrial animals usually have a protective layer (integument) which separates their internal milieu from the environment. As a result, their behavior and habits are not controlled as much by ambient chemical changes as they are by temperature and availability of water. The opposite is true of aquatic organisms. Environmental changes that belong to the realm of solution chemistry (ph changes, acid-base reactions, gas solution processes, precipitation and dissolution of solid phases, reactions of metal ions and ligands, oxidation-reduction reactions and absorption-desorption processes) greatly affect the occurrence, distribution and survival of aquatic species.

Light, which is seldom limiting in terrestrial environments, can be a controlling factor in distribution and occurrence of aquatic organisms, particularly plants. If light penetration is obscured through increased turbidity, photosynthesizing organisms cannot produce oxygen, and respiration and decay processes will quickly deplete available oxygen.

A result of these differences is that the production of organic matter is generally higher on land than in the sea. In the sea, however, efficiency of energy transfer from one trophic level to a higher is usually higher. Interruption of primary food sources in the sea may have a more immediate impact on consumers because few marine organisms have adequate energy reserves to carry them through long periods of fasting.

Differences between terrestrial and marine environments are also reflected in their response to man's activities. Effects of pollution on land are generally local unless the pollutants enter the atmosphere. Like atmospheric pollutants, aquatic pollutants are generally mobile. Their presence and effects may be observed far from the initial source of input. Man's management ability is also markedly different between terrestrial and marine environments. The historical domestication of plant and animal species on land has no marine counterpart except for limited success with a relatively few species brought into aquaculture in coastal areas.

## Classification of Coastal and Marine Environments

A number of attempts have been made to develop biogeographical classifications of marine and coastal environments. The most recent of these (Hayden et al., 1984) presents a classification for the world that combines physical processes with known biotic characteristics to obtain 40 coastal biotic provinces. This world-level scheme provides only a first approximation of environmental diversity. A smaller scale, regional approach for the United States (Table 1) developed by Ray et al. (1981) attempted to evaluate habitats both quantitatively and qualitatively for each biotic province. These schemes provide a framework which could be used

TABLE 1. PRESENCE AND IMPORTANCE OF ECOSYSTEM TYPES IN COASTAL ZONE REGIONS OF THE UNITED STATES.

| | REGION | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| ECOSYSTEM | Acadian | Virginian | Carolinian | Floridian | Louisianian | Vera Cruzian | Oregon | Sitkan | Artic | Aleutian | Indo-Pacific | Californian |
| MARITIME FOREST WOODLAND | 3 | 3 | 3 | 3 | 3 | 2 | 3 | ? | ? | ? | ? | 3 |
| COASTAL SHRUBLANDS | 3 | 3 | 3 | 3 | 3 | 3 | 2 | 2 | 3 | 3 | 2 | 3 |
| COASTAL GRASSLANDS | 3 | 3 | 3 | 3 | 3 | 3 | 2 | 2 | 3 | 2 | 2 | 2 |
| COASTAL TUNDRA | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3 | 3 | 0 | 0 |
| COASTAL CLIFFS | 3 | 0 | 0 | 0 | 0 | 0 | 3 | 3 | 3 | 3 | 1 | 2 |
| COASTAL MARSHES | 3 | 3 | 3 | 2 | 3 | 2 | 2 | 2 | 3 | 2 | 2 | 2 |
| COASTAL SWAMP FOREST | 1 | 2 | 2 | 2 | 2 | 1 | 0 | 0 | 0 | 0 | 0 | 0 |
| COASTAL MANGROVES | 0 | 0 | 0 | 3 | 1 | 2 | 0 | 0 | 0 | 0 | 3 | 0 |
| INTERTIDAL BEACHES | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 |
| INTERTIDAL MUD AND SAND FLATS | 2 | 3 | 3 | 2 | 2 | 2 | 3 | 2 | 2 | 2 | 2 | 2 |
| INTERTIDAL ALGAL ECOSYSTEMS | 3 | 2 | 1 | 1 | 0 | 0 | 3 | 3 | ? | 3 | 3 | 3 |
| SUBTIDAL HARD BOTTOMS | 3 | 2 | 0 | 2 | 0 | 0 | 3 | 3 | 3 | 2 | 3 | 3 |
| SUBTIDAL SOFT BOTTOMS | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 2 | 3 | 3 |
| SUBTIDAL GRASS BEDS | 3 | 3 | 3 | 3 | 1 | 3 | 1 | 2 | 3 | 3 | 3 | 1 |
| SUBTIDAL KELP BEDS | 3 | 0 | 0 | 0 | 0 | 0 | 3 | 3 | 0 | 0 | 0 | 3 |
| LIVING REEFS | 0 | 1 | 1 | 3 | 2 | 2 | 1 | ? | ? | ? | 3 | ? |
| PELAGIC ECOSYSTEMS | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 |

KEY: 0 = Absent 1 = Rare 2 = Common 3 = Extensive

From Ray et al. (1981)

to develop a program to conserve representative samples of each ecosystem or habitat type in a protected area system.

## Status of Marine and Coastal Ecosystems

Threats to health. Determining the state of health of the marine and coastal systems of the world is not an easy task. In many ways the problem is analogous to that of determining human health. There is no clearly defined "normal" that is applicable to every individual. Good health might be considered a condition of general well being while bad health might be considered a malfunctioning of some or all of the body. To detect, and hopefully cure ill health, it is necessary to understand structure (anatomy) and function (physiology) before pathology (disease or trauma). By analogy, detection and evaluation of marine health also requires an understanding of the function and structure of the marine portion of the biosphere.

The problem is complicated by the need to evaluate the health of the mid ocean (the main body) and the coastal waters (extremities). These coastal waters (estuaries, lagoons, fjords, bays, shallows) are the likely sites for initial system deterioration because of their proximity to the potential disruptive effects of man′s activities.

What is the health of the world′s oceans? The latest attempt to answer this question on a world scale (UNEP, 1982) reported that: in the open sea significant effects of man′s activities on ecosystem(s) have not been detected; some upward trends in contaminants have been measured, as have been some downward trends; these are not reflected in environmental deterioration; in semi-enclosed seas, shelf seas and coastal areas, however, there have been significant disturbances to ecosystems.

Numerous reports have documented the threats to coastal ecosystems in the United States. Zedler (1982) has indicated that reduction of southern California coastal marshes is so extensive that there is danger of their total elimination. The threat to these marshes continues because of their location in prime development locations. Galveston Bay, one of the most productive bays of the Texas coast, is also the one most heavily impacted by man′s activity (Shaw et al., 1981). Industry, shipping, ground water withdrawal, and oil drilling all contribute to the cumulative impact on this system which may be causing irreversible damage.

Coastal maritime forest-woodland habitats in the southeast have been severely impacted by conversion to agriculture along the Mississippi River flood plains and are threatened by tree farm monoculture and agriculture in the Carolinas and Georgia (Wharton et al., 1982). Also in the southeast, the living oyster reefs in shallow and intertidal areas are threatened by physical disturbance (dredging and excessive boat traffic), salinity changes due to freshwater diversions or hydrologic alterations, eutrophication or over enrichment from municiple waste treatment plants and urban and rural runoff and a wide range of industrial activities (Bahr and Lanier, 1981).

The mangroves of Florida (Odum et al., 1982); the subtidal grass beds of the southeast (Zieman, 1982), the coral reefs of Florida and the Gulf (IUCN, 1984); the eel grass meadows of the Atlantic Coast (Thayer et al., 1984) and the submerged vegetation of Chesapeake Bay (EPA, 1983); the high salt marshes (Nixon, 1982) and tidal flats (Whitlach, 1982) of New England all are under stresses from man′s activities in or adjacent to the coastal zone.

Our knowledge of marine ecosystems does not allow us to extrapolate these threats to specific habitats or ecosystems to a quantitative estimate of the threat to biological diversity. Intuitively the outlook is pessimistic.

Degree of protection. In reviewing a consensus that developed from the 1982 World Congress on National Parks held in Bali, Indonesia, Salm (1984) states that "marine conservation lags far behind that on land" and there have been "... few attempts to apply ecological theory to conservation management in the seas ... ".

Worldwide there are about 1200 parks and reserves that protect coastal and marine habitats (Salm and Clark, 1984). Unfortunately, the management plans for many of the coastal parks or reserves only apply to terrestrial habitats and do not extend to the adjacent aquatic ecosystems. A review of the protection offered to U.S. coastal and marine areas as part of the background studies conducted to establish policies for the Marine and Estuarine Sanctuary programs (Lynch et al., 1974) indicated that the National Park Service (NPS) managed some 46 coastal units, 22 for their historical importance, 12 for recreational value and only 12 for their natural value. The NPS estimated they only provided about one-fourth of the representation of marine or estuarine ecosystems necessary for adequate coverage of these areas.

Since that time the Department of Commerce has established 7 Marine Sanctuaries under the Marine Protection, Research and Sanctuaries Act of 1972 (with amendments). These are:

1. U.S. Monitor (1 square nautical mile southeast of Cape Hatteras, N.C. - historical purpose).

2. Key Largo (100 square nautical miles of coral reef south of Miami adjacent to Florida's John Pennekamp Coral Reef State Park).
3. Channel Islands (1,252 square nautical miles off southern California).
4. Looe Key (5 square nautical miles of a Florida coral reef tract southwest of Big Pine Key).
5. Gray's Reef (17 square nautical miles on the South Atlantic continental shelf east of Sapelo Island, Georgia).
6. Point Reyes-Farallon Islands (984 square nautical miles, off San Francisco).
7. Fagetele Bay (163 acres on Tutuila Island, American Samoa).

In addition, the Department of Commerce, in cooperation with the individual states and territories has established 16 Coastal Estuarine Sanctuaries under The Coastal Zone Management Act of 1972 (with amendments) in Maine, Massachusetts, Rhode Island, New York, Maryland, North Carolina, Georgia, Florida, Puerto Rico, California, Oregon, Washington and Hawaii.

The conservation/protection of coastal and marine habitats offered by the sites already designated under these programs is limited. The Sanctuary Programs Division of the Office of Coastal Resources Management, NOAA has developed a site evaluation list of 27 potential marine or coastal candidates for Marine Sanctuary designation (48 FR 35568, 1983). If one considers the number of common and extensive ecosystems in each of the coastal regions of the U.S. (see Table 1), it is apparent that the few estuarine sanctuaries are obviously inadequate.

Several states also have programs for protecting coastal areas. The Massachusetts Ocean Sanctuaries, California's underwater parks, Florida's

John Pennekamp Coral Reef State Park and Hawaii's Marine Life Conservation District programs are the most conspicuous of these (Lynch et al., 1974). A number of other programs such as the Experimental Ecological Reserve Network designated by the National Science Foundation, the Research Natural Area program managed by eight cooperating Federal land management agencies and the Biosphere Reserve Program provide protection for coastal and marine areas, but their principle focus is terrestrial (see National Atlas of Ecological Research Areas, USGS, USDI NAC-P-0702-75M-0, 1985). No estimates of the adequacy or completeness of habitat coverage of these several programs, either independently or collectively has been made.

## Mandates for Protection

The U.S. is party to a number of international treaties or conventions that serve to protect or conserve biological diversity either directly (by protecting threatened or endangered species) or indirectly (by protecting or preserving ecosystems). None of these deal exclusively with marine species or ecosystems. The U.S. is also party to a number of treaties dealing with fisheries (including sealing and whaling) which provide a measure of protection to species specifically named in the treaties or species contained within an area covered by the agreement. There are also a number of Federal statutes that deal specifically with marine and coastal ecosystem protection and a much larger number of federal programs that deal with conservation, natural areas or species protection in general that are relevant to marine and coastal concerns. The objectives and goals of these programs are often complex and sometimes appear in direct conflict with each other (e.g. "optimum yield" goals in fishery plans and "optimum population" goals for marine mammal protection efforts).

The principal international treaties and conventions directed towards diversity are discussed by WRI (1984). These include: the Convention on the Protection of the World Culture and Natural Heritage (the World Heritage Convention) which became effective in 1975; the Convention on Wetlands of International Importance Especially as Waterfowl Habitat (the Ramsar Wetlands Convention) effective in 1975; the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) concluded in 1973; the Convention on the Conservation of Antarctic Marine Living Resources adopted in 1980 by the Antarctic Treaty Nations; Convention on Nature Protection and Wildlife Preservation in the Western Hemisphere (the Western Hemisphere Convention) adopted in 1940; Bilateral migratory bird conventions (1916, U.S.-Canada; 1936, U.S.-Mexico; 1972, U.S.-Japan; 1976, U.S.-USSR); and the Man and the Biosphere Program of UNESCO (particularly Project 8 to establish a global system of Biosphere Reserves) initiated in 1970.

Specific domestic mandates for marine and coastal ecosystem protection are found in Title III of the Marine Protection, Research and Sanctuaries Act of 1972 as amended in 1980 and 1982, and in Section 315, Estuarine Sanctuaries, of the Coastal Zone Management Act of 1972. The former act authorizes the Secretary of Commerce with Presidential approval to designate ocean waters as national marine sanctuaries for the purpose of preserving their conservation, recreational, ecological or aesthetic values. Estuarine sanctuaries may be designated by the Secretary of Commerce, if nominated by the Governor of the State in which the area occurs "... to serve as natural field laboratories in which to study and gather data on the natural and human processes occurring within the estuaries of the coastal zone" (P.L.

92-583). Amendments to the Coastal Zone Management Act before the 99th Congress will, if adopted, change the name of the Estuarine Sanctuary Program to the National Estuarine Reserve Research System (House of Representatives Report 99-103).

More general mandates for the protection of marine and coastal ecosystems (and species) are contained in the Clean Water Act, the National Ocean Pollution Planning Act, the U.S. Fishery Conservation and Management Act, the Outer Continental Shelf Lands Act, the Coastal Barrier Resources Act, Duck Stamp Act, Fish and Wildlife Restoration Programs, Land and Water Conservation Fund Act and the various Acts creating the National Park System. The refuge system established by the funds from the Duck Stamp Act specifically protect biological diversity of waterfowl while the Land and Water Conservation Fund Act can be used for purchase of land for preservation of endangered species.

## Resource Assessment and Inventory Techniques

Resource assessment and inventory techniques used in marine and coastal ecosystems are, to be charitable, not very sophisticated. The use of modern technology, particularly involving remote techniques from satellites or airborne platforms are still in the experimental stage, although there have been successes in using aerial photography to inventory and assess trends in wetlands (Hardisty and Klemas, 1983) and submerged vegetation in coastal areas (Orth and Moore, 1981). Techniques for compiling and presenting existing information have become more sophisticated in recent years, particularly with increased use of computers to store screen and present data in a synthesized format (NOAA, 1984).

The basis for most of the current inventories or resource assessments are surveys using traditional collecting gear such as nets, dredges, coring devices, grabs, water sampling equipment, and direct observation, sometimes supplemented with photography or video recording. Sampling platforms are usually surface ships, but more use is being made of subsurface vessels (either manned submersibles or remote operated vehicles) and aircraft.

The traditional techniques are limited in several ways. Most of the equipment is sampled "blind", i.e. we cannot see exactly how the equipment is performing or, more importantly, what is not being sampled by the equipment. Most sampling equipment is also relatively indiscriminate. It picks up everything which cannot avoid or escape it, within its size constraints. Processing samples taken in this fashion can sometimes be very tedious, labor intensive and expensive. There is also frequently a problem with identification of many of the taxa found in marine and coastal areas.

In recent years we have been able to enhance our assessments based upon use of traditional sampling by adding a visual component with photography or video (or even direct observation from submersibles or by divers). The restricted range of vision underwater, however, limits our ability to accurately estimate a population's or habitat's areal distribution or configuration particularly in comparison to our ability to do so on land.

The data base for resource assessment is most complete in the area of fisheries management. Extensive field efforts have been mounted by the Federal government, particularly the National Marine Fisheries Services, and state fisheries resource or research agencies to collect data for the purpose of stock assessment. Assessments based upon field efforts can be evaluated in subsequent years through comparison with commercial and recreational fisheries landings when available. This information has, however, limited use in evaluating biological diversity and ecosystem changes except as reflected in fish species susceptible to the gear used in the surveys.

The first attempts to develop a comprehensive picture of U.S. coastal and marine areas began in response to Federal environmental legislation in the late 1960's. As a part of the National Estuarine Pollution Survey, Odum, Copeland and McMahan (1974) prepared a comprehensive 4 volume report on the Coastal Ecological Systems of the United States. This report was the first attempt to organize the knowledge of estuaries by type and identify the types in specific regions.

A more ambituous effort was initiated with respect to coastal and outer continental shelf areas with potential for oil and gas development by the Bureau of Land Management (now Mineral Management Service - MMS) in 1973 (Monastero, 1976). Socio-economic and environmental inventories of existing information were compiled, reconnaissance field studies were conducted and benchmark data for selected areas were established. The biological characterizations conducted under this program were some of the most thorough conducted on outer continental shelf areas. Unfortunately, the initial characterizations have not been followed up with continuing monitoring by MMS or any other agency in any consistent or regular fashion.

In 1974 the U.S. Department of Interior established the Office of Biological Services to strengthen U.S. Fish and Wildlife Service's ability to predict and accurately assess the impact of land and water developments on fish and wildlife. Under this program a number of coastal characterizations have attempted to develop a regional synthesis of environmental data. In 1979 the National Coastal Ecosystem Team initiated a series of community and estuarine profiles which synthesizes scientific information regarding critical coastal habitats or ecosystems (Pendleton and Kitchens, 1984). The profiles developed under this program (many of which have been cited in this paper) provide a valuable resource for the planning of a program to protect coastal and marine ecosystems.

Comprehensive approaches to resource assessment and inventory of marine and coastal regions are being developed by NOAA's Ocean Assessment Division (OAD). Since 1979, they have been organizing and synthesizing the best available information on selected characteristics of the Exclusive Economic Zone (EEZ) and conducting comprehensive, strategic assessments useful for reducing potential conflicts inherent in multiple use of resources (Ehler and Basta, 1984). This program is presently, in cooperation with the National Marine Fisheries Service, collecting and synthesizing information on the spatial and temporal distribution of the life history stages of about 300 species of fishes, invertebrates, reptiles and mammals. They are also developing compatible data bases on pollutant discharges in the coastal zone and EEZ and on physical characteristics of the EEZ including sea surface temperature from NOAA satellites. Another project is developing a computer based national estuarine inventory which will contain physical and hydrologic boundaries as well as selected physical, biological and economic characteristics of at least 90% of the nations large and medium sized estuaries. A Status and Trends program in the OAD composed of mussel watch, benthic surveillance, water quality, and historical data base development will enable the quantification of the current status and the monitoring of long term temporal and spatial trends of key contaminants, water quality parameters and biological indicators of environmental quality in estuarine and coastal waters.

Resource assessment and inventory techniques have evolved rapidly over the past several years, as federal effort is dedicated to these areas. The results of these activities are still primarily descriptive and retrospective although quantification and timeliness is improving with increasing use of computers and near real time data input from satellites and in-situ instrumentation. The present state of the art is sufficient to plan a system to maintain marine and coastal ecosystem diversity. There still exists, however, a major gap in our knowledge of the distribution and abundance of non-commercial species. Present resources available for assessment and inventory activities will not support comprehensive biotic inventories. Because of traditional lack of support for activities of this nature, there is also a shortage of trained systemetists for many taxa.

Research is underway on a number of fronts to provide us with better assessment techniques. There is some indication that automatic image analysis techniques can be developed which will enable rapid and accurate quantification and identification of plankton. Much more research needs to be conducted before such techniques will be ready for routine use. Some success with acoustic censusing of fish populations has been obtained. The best success has been in counting single or relatively small numbers passing a sampling point such as during migrations up a river or through a fishladder or fishway. The major impediments to improved resource assessment methods are both technical and conceptual. We lack good, verifiable models which can convert real time information on environmental factors into predictable impacts on known resources. We also lack the technical means to rapidly assess the present status of some stocks of interest.

There do not appear to be any environmental, social/institutional or political barriers to improving the state-of-the-art of resource assessment if economic resources are available. Unless the need for improved resource assessment is better understood throughout all segments of the population, it is unlikely that this area will be perceived as having a high enough priority to obtain a greater proportion of limited national resources. The only way to overcome this problem is through better understanding of the problem by resource managers and more importantly the public.

## Strategies Used to Identify and Establish Areas for Maintenance of Biological Diversity

At the present time, maintenance of biological diversity, *per se*, is not the principle consideration in identifying and establishing protected marine areas. Most of the theoretical framework for selection strategies involves identifying ideal, representative habitats within each coastal province. This theoretical framework quickly runs afoul of the reality of politics. What is available and/or attainable frequently weighs more heavily in the establishment process than the theoretical considerations used to identify a number of potential sites.

The National Park Service conducted a systematic study under its National Landmarks Program (Sudia, 1974) which was designed to determine which areas merited consideration for national landmark status or ultimately for inclusion in the National Park System. This program considered coastal and marine landforms (seashores, islands, coral islands, reefs and atolls) and aquatic ecosystems (marine environments and estuaries) as themes within the coastal natural regions identified within the U.S. The process employed by the NPS attempts to obtain representation of all themes important to each natural region.

The National Marine Sanctuary Program originally developed a classification of coastal regions based primarily on Ketchum (1972) and invited nominations for areas to be placed on a list of Recommended Areas, with justification based upon the regional classification and other criteria established by regulation. The List of Recommended Areas (44 FR 44831, 1979) was supposed to serve as a catalog of potentially significant marine sites for consideration for Marine Sanctuary status. The List of Recommended Areas criteria were broad and allowed marginally acceptable nominations to qualify. Much controversy arose over the entire program. In June, 1983 (48 FR 24296) the List of Recommended Areas was eliminated and replaced with a Site Evaluation List process detailed in the program development plan (NOAA, 1982). This process involved the convening of eight regional teams of scientists, broadly knowledgable of the ecosystems in their region, who reviewed lists of recommended sites already developed from a variety of sources and, where appropriate, selected additional sites. The teams used a resource classification system in the program development plan which incorporated zoogeographic regions, coastal biotic provinces, resource value and human use value. Preliminary lists developed by the regional teams were circulated widely for public and agency comments. Based upon public comment and selection criteria established by NOAA (40 FR 24296, 1983) a list of 4 to 5 sites per region was submitted to NOAA, which after a period of public comment promulgated a Site Evaluation List of 27 marine and coastal sites (48 FR 35568, 1983). This list will serve as the source list for the selection of active candidates for Marine Sanctuary status for the next several years, at which time the list will be reviewed.

The Estuarine Sanctuary Program has also established a biogeographic classification framework (27 subregions within 11 biogeographic regions, including the Great Lakes). The eventual goal is to establish estuarine sanctuaries which represent typologies found within each of the 27 subregions. There is no active program or strategy to identify potential candidates at the Federal level. States interested in participating in this program undertake a site selection process, and nominate the site or sites selected to NOAA for review. If the site meets the criteria established by NOAA (49 FR 26502, 1984) the site can be designated a National Estuarine Sanctuary.

The Man in the Biosphere Program is using a process similar to that used to establish the site evaluation list for the Marine Sanctuary Program to select marine and coastal candidates for Biosphere Reserve designation. Regional *Ad hoc* Selection Panels of scientists (and resource managers) are assembled to review lists of natural areas and, based upon their knowledge of the regions, suggest additional areas to be evaluated. The panels utilize the process and criteria developed by Ray et al. (1981) to develop recommendations to the U.S. Man in the Biosphere Directorate on Biosphere Reserves, which then has the responsibility for shepharding approved sites through the designation process.

Utilization of modified Delphic or nominal group processes to initially identify potential sites, followed by detailed assessment of the sites as does the marine sanctuary program and the Man-in-the-Biosphere program, should result in the identification of superior sites from a scientific and technical perspective. The weakness in this process is that final selection and design of a sanctuary is subject to the political, economic and social constraints which may result in the final area designated being far from optimal.

It is difficult to exert control over marine and coastal areas in the same way as can be exerted over terrestrial sites. Marine and coastal sites even when protected, retain a vulnerability to impacts arising both within and outside of the site´s boundaries. Care should be taken that the core or specific features that are the raison d´etre for identification are protected by an adequate buffer to minimize the potential for impact from outside of the protected area. Wherever possible, incorporation of a buffer should be implicit in the design of a marine or coastal protected area.

## Strategies Used to Manage Marine/Coastal Systems for Biological Diversity

Lynch (1974) developed four classifications or types of marine and coastal natural areas around which management strategies could be developed. Habitat preserves, which might be established to preserve, protect or manage representative habitats of outstanding quality (or essential or specialized habitats utilized by rare or endangered species), would usually require some measure of total preservation requiring severe use restrictions.

Species preserves which would be established to conserve specific genetic resources might not need such severe constraints as habitat preserves. Management strategies for such areas must be flexible enough to adequately protect the target species during particularly vulnerable portions of the life cycle such as spawning. An orientation towards species preservation might require protection of migratory pathways, spawning grounds, nursery grounds or feeding grounds separated in both time and space.

A third category of preserves, those established primarily for research, could give rise to two conflicting strategies for management. Research and educational activities might be restricted to nondestructive observation and data collection as is practiced in Federal Research Natural Areas or manipulative studies might be sanctioned as is done in some Experimental Ecological Reserves or Biosphere Reserves. It is difficult to reach a consensus even among scientists as to the degree to which manipulative studies should be sanctioned, particularly in fragile coastal areas which are already under stress from many sources. There is a school of applied scientists that believe field laboratory areas in which modifications up to and including species removal or severe modification (destruction) may be necessary to adequately study impacts of stress and/or restoration processes (Cronin, 1974). Recreational or esthetic preserves established primarily to provide for public enjoyment could employ a large variety of strategies. A management plan emphasizing maximum active recreational use of shore and shallow aquatic areas will differ in almost all respects from a plan to provide pleasures associated with a wilderness type experience.

Each of these broad strategies has been used to some extent or another in marine and coastal areas. With the exception of the USS Monitor Sanctuary, all of the marine sanctuaries presently established have habitat protection as a principle goal, although multiple use is allowed where such use is not expected to adversely impact on the resources being protected.

The Commonwealth of Virginia has established a blue crab "sanctuary" in lower Chesapeake Bay during the prime spawning season for blue crabs in which crabs may not be taken commercially. This serves as a "species preserve" during a critical time of the blue crab life history. Unfortunately no studies have been conducted to evaluate the contribution of the presence of this "sanctuary" to Chesapeake Bay blue crab stocks.

Several coastal Research Natural Areas have been established in coastal areas. No assessment has been made to determine the extent to which the Research Natural Area system provides coverage of U.S. coastal and marine habitats. About fourteen Experimental Ecological Reserves are located in coastal areas. The extent to which manipulative research is conducted in these areas has not been assessed. One site, the Smithsonian Institution, Environmental Research Laboratory in Maryland, does conduct manipulative research on a small watershed draining into an estuarine tributary to Chesapeake Bay.

Many coastal preserves are managed primarily for recreation. The principle examples are, of course, the National Seashores managed by the National Park Service. The management plans for the National Seashores are tailored to the specific seashore, but to the maximum extent possible considering their recreational focus, include provisions for natural resource protection up to including designation of portions of the seashore as National Wilderness Areas.

The National Marine Sanctuary Program (NOAA, 1982) has set forth a number of concepts related to size and zoning for protecting marine and coastal areas based upon the work of Ray (1975). These concepts revolve around the identification of a "core" area or areas embodying the values the sanctuary is designed to protect. The "core" area should be ideally protected by a buffer, an area adjacent to or surrounding the core within which management practices appropriate to protecting the buffer can apply. Present management plans for marine and estuarine sanctuaries, most of which were developed, or at least initiated, prior to promulgation of the Program Development Plan do not, however, directly reflect these concepts.

Unfortunately, we were not able to evaluate success of a specific concept or strategies for maintenance of biological diversity because we were unable to identify maintenance of biological diversity as a specific goal of any marine or coastal protected area. In fact, there is some indication that designation of some areas as protected areas has had a negative impact on the natural resources of the areas because of increased public attention, resulting in increased visitation.

The strategies and concepts of management available to managers of protected coastal and marine areas, however, incorporate theoretical considerations related to biological diversity (see Salm and Clark, 1984). Given a mandate to protect biological diversity, concepts such as habitat preserves and species preserves (Lynch, 1974) and techniques such as core area identification and designation with buffer area protection (Ray, 1975; NOAA, 1982) applied across the full gamut of marine and coastal environments (Hayden et al., 1984; Ray et al., 1981; NOAA, 1982) would provide a strong program to protect and maintain marine diversity.

## Protection of Genetic Diversity in Marine and Coastal Species

Traditional practices for protecting or preserving genetic diversity in specific terrestrial plant and animal species include: establishment and maintenance of natural ecosystems (e.g. parks and other natural areas);

storage and banking of germ plasm; and establishment and maintenance of botanical gardens and zoos. With the exception of the establishment and maintenance of natural ecosystems (which has been discussed in context of ecosystem diversity), programs to protect or preserve genetic diversity in marine or coastal species are essentially non-existent.

Very few studies of storage and banking of genetic material in marine and estuarine species have been conducted. Spermatazoa and eggs and larvae of some echinoderms (Dunn and Machachlan, 1973; Asahina and Takahashi, 1978), spermatazoa of oysters (Zell et al., 1979; Hughes, 1973; Lannan, 1971), and spermatazoa of _Limulus_ (Behlmer and Brown, in press), have been successfully cryopreserved in laboratory studies. No organized depository for genetic material for any of these organisms has been established. (Some marine and coastal phytoplankton species, however, have been deposited in algal type collections, but the majority of algae in these collections are fresh water.)

Aquaria have not filled the role with respect to protection of genetic diversity of aquatic organisms that has been filled by botanical gardens and zoos with respect to terrestrial flora and fauna. The ornamental fish industry, which is primarily organized around tropical freshwater species, but which does include some marine species, is probably an exception to the almost universal lack of programs to protect, enhance, maintain or develop genetic diversity.

Although Ryther (1981) estimated the worldwide 1971-78 production from mariculture (aquaculture of marine organisms) to be just under 7 billion pounds, no organized program for preservation of germ plasm of key species has been established. Mariculture is presently restricted primarily to fishes, molluscs (oysters, clams, scallops, mussels, abalone), crustaceans (shrimp, lobster), bait worms and seaweed. Lack of knowledge of genetic characteristics of commercial species and gene pool maintenance is considered a major impedement to full realization of the mariculture potential in the U.S. (JSA, 1983).

## Recommendations for Improvements in the Ability to Maintain Biological Diversity in Marine/Coastal Environments

The United States presently has available to it a number of institutionalized mechanisms to establish protected areas in marine and coastal regions. These have been discussed in earlier sections of this paper. It is obvious, from analysis of these institutional mechanisms, that they are not being used to develop a comprehensive national program for protection of marine/coastal biological diversity.

The single most important factor in the failure to develop such a comprehensive national program is lack of concern for the problem. This lack of concern results in a low national priority for the specific issue of marine/coastal diversity, which translates into limited funding for either research to develop technologies useful in assessing marine/coastal biological diversity or for identification and establishment of areas designated for the purposes of maintaining or protecting biological diversity.

In the absence of a ground swell of public concern, encouragement and support for activities in this area must come from the Executive or Legislative leadership. In this era of deficit budgets and top Executive occupation with crisis management, the initiative and leadership must come from Congress. The benefits of protecting marine/coastal genetic and ecological diversity are not as readily apparent as are benefits deriving from comparable activities in terrestrial environments. The old saying "Out of sight, out of mind" is particularly true about marine/coastal problems.

This inability to view impacts (or even the resources themselves) is exacerbated by the difficulties with resource assessment in marine and coastal areas, particularly those that occur subaqueously. Even those proponents of a national program to protect biological diversity usually have a terrestrial orientation. Statements justifying a program to protect biological diversity because of its necessity to improved agriculture, future medical discoveries or the raw material for industrial innovations seldom include references to mariculture, drugs from the sea or colloids from seaweeds.

A number of general and specific actions can be taken by Congress in its oversight role to ensure recognition of the legitimate needs for protection of biological diversity in marine/coastal environments. These include: 1) setting requirements that terrestrial oriented areas established for protection of natural resources specifically recognize and address marine/coastal ecosystems that abut or are enclosed by their boundaries; 2) setting requirements that management plans for all marine/coastal areas established for protection of natural resources specifically address protection of biological diversity; 3) establishment of an interagency task force or coordinating mechanism to provide a periodic combined assessment of the quantity and quality of protected marine/coastal ecosystems; and 4) specific encouragement of Federal funded research programs to support research in improved resource assessment technology, innovative information synthesis and display and development of improved conceptual and quantitative models for understanding and management of marine/coastal ecosystems.

The most promising emerging technologies for resource assessment can be classed in an overall category of image processing. The images may range from global or near global satellite generated images, acoustic images from sonar or similar equipment, aerial photography, video imagery linked with microscopy, or any number of variations or combinations thereof. The processing involves the use of modern computer, information analysis, and presentation technology. Together, these provide us with a potentially very powerful tool for resource assessment and management once the calibration and verification of the methodologies are acceptable.

It is also essential that marine/coastal protected area management recognize the differences between these areas and terrestrial protected areas. The necessity for understanding fundamental characteristics of marine ecosystems and the difficulty of conserving or managing to protect diversity on a species-by-species level is critical. We must develop improved conceptual models for the interactions of ecological processes and individual species in these environments. Where possible these conceptual models should also be expanded into quantitative models to assist coastal management, but the development of conceptual models is necessary to provide the foundation for improved public understanding and defensible management approaches.

The Congress could also direct (and provide appropriate resources to effect) that the Marine Sanctuary program proceed more rapidly to incorporate high quality areas representative of all identified habitats within each coastal province within its system. The Estuarine Sanctuary program could also be directed to develop a more active Federal role in site identification until a representative of each habitat is incorporated into the system. The mandates for these two programs should be expanded to specifically reflect their potential importance in a national program to maintain and protect marine/coastal biological diversity.

Public understanding and concern is the key to successful development of a national program for the maintenance of marine/coastal productivity. Public receptivity of the need to protect coastal areas is being amply demonstrated by the wide popular support for the Federal and State efforts towards the cleanup of Chesapeake Bay (Lynch, 1985) and interest in coastal areas in general (Fegan, 1985).

At this time new legislative programs to provide for the protection or maintenance of biological diversity in marine/coastal ecosystems do not seem warranted. The present Federal programs available for this purpose can, with focused oversight and minor modifications, provide a framework for a national program for protection of marine/coastal diversity.

Strong programs are needed to enhance public awareness of the problem and educate the public on the unique problems associated with marine/coastal area and diversity protection. Without such educational programs, it is unlikely that programs for the protection and maintenance of marine/coastal biological diversity will receive a high enough National priority to be fully implemented.

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Contract Paper No. 7

# Federal Laws and Policies Pertaining to the Maintenance of Biological Diversity on Federal and Private Lands

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Environmental Defense Fund

October 1985

Office of Technology Assessment
U.S. Congress

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# FEDERAL LAWS AND POLICIES PERTAINING TO THE MAINTENANCE OF BIOLOGICAL DIVERSITY ON FEDERAL AND PRIVATE LANDS

a report by

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of the

Environmental Defense Fund

to the

Office of Technology Assessment

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# ABSTRACT

The conservation of biological diversity is directly affected by a host of federal laws, regulations, and programs, though it is an explicitly stated objective of virtually none of these. Among the federal laws pertaining to the conservation of biological diversity are several that have species conservation as their purpose and many others that have the preservation of natural areas as their purpose. Of the former, the Endangered Species Act is the most comprehensive and important for biological diversity generally.

Natural area preservation is the principal mandate of certain federal land systems and but one of several, often conflicting, mandates for other federal land systems. Private natural area preservation, or at least conservation of natural resource values on private lands, is the objective of several federal laws, principally those pertaining to agricultural lands.

Effective biological conservation efforts of federal agencies can be enhanced by greater coordination of the natural areas programs with the species conservation programs. In particular, the extensive lists of "candidate" species identified for future protection under the Endangered Species Act provide a highly useful guide for directing natural areas programs so as to maximize their usefulness to biological diversity conservation goals.

## Introduction

This paper identifies and assesses federal laws and policies pertaining to the maintenance of biological diversity on federal and private lands in the United States. It is important to note at the outset that although the concept of "biological diversity" and the importance of its conservation are now generally recognized by the scientific, and much of the lay, communities, the term itself has virtually never been used in federal legislation. The only exception, apparently, is in Title VII of the Foreign Assistance Act of 1983, 22 U.S.C. 2151q, which directs the Administrator of the Agency for International Development to develop a "strategy ... to protect and conserve biological diversity in developing countries." The notion of preserving "diversity" as an explicit objective of federal law also appears in the general policy declarations of the National Environmental Policy Act, 42 U.S.C. 4321 et seq.. There it is declared to be the policy of the federal government to "preserve important historic, cultural, and natural aspects of our national heritage, and maintain, wherever possible, an environment which supports diversity and variety of individual choice." Id. at 4331(b)(4). Elsewhere, however, one searches in vain for any explicit federal legislative directive to conserve biological diversity.

Despite the paucity of federal laws that specifically refer to biological diversity, there are obviously a great many that directly or indirectly affect biological diversity and its conservation in a very substantial way. Some of these are rather obvious, like the Endangered Species Act, 16 U.S.C. 1531 et seq., which declares its purposes to be "to provide a means whereby the ecosystems upon which endangered species and threatened species depend may be conserved [and] to provide a program for the conservation of such" species. Id. at 1531(b). Others are less obvious but arguably no less important. The Clean Air Act, 42 U.S.C. 7401 et seq., for example, mentions wildlife only once, in its definition of "public welfare." Id. at 7602(h). The protection of public welfare, in turn, is the object of the so-called "secondary ambient air quality standards" required to be promulgated for various air pollutants. The preservation of biological diversity in large areas of the Northeast, the Rocky Mountains, and elsewhere is now clearly linked to the levels of emissions of sulphur and nitrogen oxides. Thus, the Clean Air Act, though it scarcely mentions wildlife, and then only in connection with its secondary objectives, may be as important for the conservation of biological diversity as any of the other laws that purport explicitly to be concerned with conservation.

In light of the above, one could treat all of environmental law, and even a good many non-environmental laws, in an assessment of federal legislation pertaining to the maintenance of biological diversity. In a paper of only thirty pages, however, one could scarcely say more than a paragraph about each. This paper, therefore, will focus more narrowly on those federal laws and programs that are implemented for the primary purpose of conserving wild plants and animals and natural areas. In short, its focus is on those that are concerned with the preservation of our nation's natural heritage. Even so limited, the treatment here will be necessarily cursory. A decade ago, the Nature Conservancy prepared, and the National Park Service published, a comprehensive report on federal laws and programs pertaining to the preservation of natural areas. That report, Preserving Our Natural Heritage, took the form of a more than 300-page book. This paper attempts to cover most of the same ground as well as a decade of new developments in a tenth of the space.

## Species Conservation Laws

There are a fairly limited number of federal laws that have the conservation of particular species as their express object. Historically, authority to conserve and manage wild animals was asserted initially by the states and still rests for the most part with the states. Federal assertions of authority in this area have been limited and infrequent. Neither the federal nor state governments have taken much initiative for the conservation of wild plants until quite recently.

### 1. The Endangered Species Act

Clearly the most general and important of the federal species conservation laws is the Endangered Species Act of 1973. A detailed explanation of the law and its implementation can be found elsewhere. (Bean, 1983) For present purposes, a summary description of some of its key features will suffice. Authority for the implementation of the Act is shared by the Secretaries of Interior and Commerce. The former, who has delegated his responsibilities to the United States Fish and Wildlife Service, is responsible for terrestrial and freshwater animals and all plants. The latter, operating through the National Marine Fisheries Service, is responsible for marine animals. The basic responsibilities of these agencies are to list species as endangered (currently in danger of extinction) or threatened (likely to be in danger of extinction within the foreseeable future) and to protect them. Currently, some 277 animals and 75 plants that occur in the United States are listed.

The most basic purpose of the Act is to prevent the extinction of listed species. Achieving the recovery of listed species to the point that they are no longer endangered or threatened is the Act's ultimate objective. The means by which the Act seeks to attain these goals are restrictions on the taking of protected species and on commerce in them. See 16 U.S.C. 1538. In addition, Section 7 of the Act obliges all federal agencies to insure that their actions not jeopardize the continued existence of any listed species or adversely affect its critical habitat. Id. 1536. "Critical habitat" is a concept introduced by the Act that has a somewhat specialized meaning. It refers only to those areas that have been officially designated, as a result of a rulemaking procedure, as "essential to the conservation of the species." Id. 1532(5). The only functional significance of a critical habitat designation is through the federal agency action command of Section 7.

Section 7 has had a major impact in stimulating greater attention to the ecological consequences of planned actions by agencies not otherwise responsible for species conservation. Regular and frequent communication between the Fish and Wildlife Service or National Marine Fisheries Service and such other agencies is the key to compliance with Section 7, and compliance is a very high priority for most agencies. The reason, quite simply, is the provision the Act makes for citizen initiated lawsuits to enjoin violations of the Act, including violations of Section 7. Thus, agencies take Section 7 quite seriously because they know that their planned activities can be put in jeopardy if Section 7's commands are not met.

In addition to fostering closer cooperation among federal agencies, the Act has also fostered cooperative approaches between state and federal agencies. The Endangered Species Act was passed in large measure because of a congressional perception that the states had not demonstrated a commitment to the preservation of imperiled species that was commensurate with the magnitude of the problem. Rather than merely displace the states from any authority over endangered and threatened species, however, the Act sought to stimulate action by the states to develop their own parallel and complementary endangered species programs. As an inducement to do so, the Act offered the opportunity to receive financial aid for the implementation of state programs that met certain requirements specified in the Act. Despite the fact that the federal monies available have been modest and irregular, most states have in fact responded by enacting their own parallel endangered species laws (currently 42 states have endangered animal programs that qualify for federal assistance and 17 have qualified plant programs). As a result, the accomplishments of the Endangered Species Act cannot be discerned by examining the activities of federal agencies alone.

Private efforts aimed at species conservation have also been facilitated by the Endangered Species Act. In addition to the already mentioned right to enforce the Act through citizen lawsuits, private individuals or organizations can participate in its implementation in various other ways. One that has been extensively used is the opportunity to petition the appropriate Secretary to list a particular species as threatened or endangered. Currently, about 3,000 United States species are administratively designated as "candidates" for future listing action; a substantial number of these were the subject of citizen initiated listing petitions.

Although the Endangered Species Act does not specifically require the development of a biological resource inventory or data bank, the duties to list any plant or animal species that meets the Act's definitions and to consult with other federal agencies about the impacts of their planned actions on listed species create an obvious need to compile an extensive amount of information about the abundance and distribution of not only listed species but also those under consideration for future listing. In an effort to systematize the collection of data for the latter, the Fish and Wildlife Service prepares and periodically publishes comprehensive lists of its formal "candidates" for future listing action. These notices have enabled federal land managing agencies on whose lands these species occur to take them into account when making land use decisions even before the species is formally listed. They have also served to influence the acquisition priorities of land acquiring private conservation organizations like the Nature Conservancy. As a result of such private and federal agency initiatives, the urgency to list certain of the candidate species has lessened, and the need to list others may be eliminated.

For at least a few of the candidate species, neither listing nor alternative protection has occurred in time. As many as eight such species are believed to have gone extinct within the last three years. That fact has underscored the need to improve the monitoring of the status of candidate species so as to take listing or other action in time to prevent their loss. Legislation now pending in Congress would require the Secretary to monitor more carefully the status of candidate species. It should be acknowledged, however, that the federal government's ability to monitor the status of even the relatively small number of listed species is limited. Within the last eighteen months, the first extinction of a listed species, the Palos Verdes blue butterfly of California, is believed to have occurred. For most of the other listed species in the United States, we simply do not yet know whether they are increasing, declining, or remaining constant. Thus, absent a significant strengthening of the federal endangered species program, principally consisting of increasing the resources available for its implementation, it alone will be unlikely to prevent the further decline and possible extinction of many species already known to be in trouble.

## 2. Other Species Conservation Laws

Other federal species conservation laws are less sweeping in both their scope and their objectives. The oldest of these, the Lacey Act of 1900, 16 U.S.C. 3371 et seq., enlists the federal government in the enforcement of state wildlife and (at least since 1981) endangered plant conservation laws when violators of those laws engage in interstate commerce. Although the Lacey Act is thus derivative in nature, deriving its substance from the laws of the states, it provides a significant law enforcement supplement to the efforts of states to implement their own laws pertaining to the conservation of biological diversity. Because of the very modest penalties that attached to Lacey Act violations, however, it was not a very effective deterrent for most of its history. Since 1981, when the penalties were substantially increased, the Act has become a more formidable tool for protecting biological diversity.

The oldest direct federal species conservation law is the Migratory Bird Treaty Act of 1918, 16 U.S.C. 703 et seq. Enacted to implement a 1916 treaty between the United States and Canada, it reflected in part a reaction to the then recent extinction of the passenger pigeon and the plundering of many plume birds for the millinery trade. Although the Act applies to all migratory birds, including both game and nongame species, by far the greatest share of affirmative management efforts goes to waterfowl and other migratory game birds. Those affirmative management efforts take two principal forms: comprehensive regulation of the hunting of birds and the establishment and operation of an extensive system of migratory bird refuges to protect key breeding, resting, and wintering areas. The authority for the latter was initiated by the Migratory Bird Conservation Act of 1927, 16 U.S.C. 715 et seq. The migratory bird refuges comprise a substantial part of the National Wildlife Refuge System, about which more will be said later. For present purposes it is important to note that the acquisition of wetland areas for waterfowl conservation is closely coordinated between the state and federal governments and is aided substantially by the private acquisition efforts of Ducks Unlimited and other groups. In general, the federal migratory bird program has been successful in maintaining fairly stable populations of waterfowl. Certain species, among them various geese of the Pacific flyway and black ducks, have experienced recent, marked declines. In addition, many nongame migratory birds have seriously declined.

The federal program for the conservation of migratory game birds is not directly concerned with biological diversity in the sense of preventing the extinction of species. Rather, its objective is to perpetuate an abundant resource for continued recreational hunting. The widespread system of refuges that has been established for this purpose, however, clearly benefits many other life forms associated with the protected wetland habitats.

The Bald and Golden Eagle Protection Act, 16 U.S.C. 668 et seq., the Marine Mammal Protection Act, 16 U.S.C. 1361 et seq., and the Wild, Free-Roaming Horses and Burros Act, 16 U.S.C. 1331 et seq. are other, more limited, federal species conservation laws. Each of these statutes was enacted in response to a perceived threat of serious depletion of the subject species. All predate the enactment of the much more comprehensive Endangered Species Act. The principal conservation tool each employs is simply a restriction on the taking of the protected animals. Other federal conservation laws that would warrant treatment in a longer discussion of biological diversity are identified in Table 1.

Table 1

Other Federal Wildlife Conservation Laws

Airborne Hunting Act, 16 U.S.C. 742j et seq.
Anadromous Fish Conservation Act, 16 U.S.C. 757b
Antarctic Marine Living Resources Act, 16 U.S.C. 2431 et seq.
Dingell-Johnson Act, 16 U.S.C. 777 et seq.
Fish and Wildlife Conservation Act of 1980, 16 U.S.C. 2901 et seq.
Fishery Conservation and Management Act, 16 U.S.C. 1801 et seq.
Fur Seal Act, 16 U.S.C. 1151 et seq.
National Aquaculture Act, 16 U.S.C. 2801 et seq.
Pacific Northwest Electric Power Planning and Conservation Act, 16 U.S.C. 839 et seq.
Pittman-Robertson Act, 16 U.S.C. 669 et seq.
Salmon and Steelhead Conservation and Enhancement Act, 16 U.S.C. 3301 et seq.
Sikes Act, 16 U.S.C. 667a et seq.
Whaling Convention Act, 16 U.S.C. 916 et seq.

In summary, federal species conservation laws provide a useful framework for addressing certain threats to biological diversity, most notably those that arise from overexploitation and commercial trade. With the exception of the Endangered Species Act, few address in a significant way the generally more serious threats to diversity arising from the alteration of natural habitats. To meet these threats, various natural area protection laws have been enacted.

## Natural Area Protection Laws

### 1. Special Use Land Systems

Natural area protection, and with it the conservation of the biota associated with such areas, is a major or minor goal of several federal statutes establishing federal land use systems. The National Park System is the oldest of these, consisting of vast and diverse areas set aside for the "the fundamental purpose of . . . conserv[ing] the scenery and natural and historic objects and the wildlife therein and .. provid[ing] for the enjoyment of the same in such manner and by such means as will leave them unimpaired for the enjoyment of future generations." 16 U.S.C. 1. The National Park Service, which administers the National Park System, is thus responsible for fulfilling two goals that have at least the potential for conflict: the preservation of natural resources and the promotion of their enjoyment through public recreational use. A more serious and only recently recognized problem is the potential for impingement on both of these goals from uses of adjacent and nearby lands. Many western parks are bordered by Forest Service or Bureau of Land Management lands managed for multiple-use purposes, including timbering, energy development and other intensive uses, the effect of which may be detrimental to park resources. Though both Congress and the Executive branch are now seeking better ways to secure cooperation among adjoining land management agencies, the failure thus far to assure that land use decisions on federal lands adjacent to national parks do not run counter to park resource conservation efforts is one of the more serious obstacles to securing the full potential of the parks to contribute to the conservation of biological diversity.

Conservation of living natural resources is the principal mission of the National Wildlife Refuge System, administered by the United States Fish and Wildlife Service. Whereas the National Park Service, at least in its so-called "natural parks", seeks to achieve its conservation objectives by allowing natural forces to work, the Fish and Wildlife Service more often seeks to manipulate those forces so as to produce desired results in terms of abundance of some specific wildlife resource. The basic legislative authority governing the refuge system is the National

Wildlife Refuge System Administration Act, 16 U.S.C. 668dd et seq. Though it provides some very generally stated policies applicable to the purposes and management principles for the Refuge System as a whole, there is also considerable diversity as to the objectives that each refuge in the system is to serve. As previously noted, many refuges have been included in the system for migratory waterfowl conservation purposes. These are often intensively managed to manipulate water levels, vegetation, and other characteristics. Many other refuges have been acquired for the purpose of preserving endangered species found there; these may or may not require intensive management for that purpose. Still others have been established to conserve other specified game or non-game animals. Because all uses of refuges other than wildlife conservation are secondary, it represents a major federal land system through which the conservation of biological diversity can be advanced.

### 2. Multiple Use Land Systems

By far the most extensive federal land systems are those managed for multiple use purposes by the Agriculture Department's United States Forest Service (the National Forest System) and the Interior Department's Bureau of Land Management (the National Resource Lands). As previously noted, the management of these lands can have adverse effects on the natural resources of adjacent lands managed for conservation purposes. At the same time, however, federal multiple-use lands offer many opportunities to advance the conservation of biological diversity. Under the National Forest Management Act and the Federal Land Policy and Management Act, the Forest Service and BLM, respectively, must maintain inventories of resources -- including the living resources -- of the lands under their jurisdiction and manage those lands for purposes that include wildlife conservation. Despite their multiple-use mandates however, the two agencies have principally been oriented to commodity production and have given relatively little attention to conservation.

Two historically parallel efforts, one statutory and the other administrative, illustrate two quite different approaches to the task of conserving biological diversity on federal multiple-use lands. In 1964, with passage of the Wilderness Act 16 U.S.C. 1131 et seq., Congress superimposed a new land management system, the National Wilderness Preservation System, on three pre-existing systems, the National Forest, National Wildlife Refuge, and National Park Systems (BLM lands also became eligible for inclusion in the wilderness system as a result of the Federal Land Policy and Management Act). The new law did not change agency jurisdiction over any land areas, but it did impose very stringent restrictions on how agencies were to manage areas formally designated as wilderness. Wilderness areas were to be designated from among the various systems' essentially pristine areas, areas of 5,000 acres or more that were roadless and without other substantial evidence of human alteration. Once designated, they were to be managed so as to retain their natural, wilderness character by prohibiting the construction of roads or structures and the use of motorized transportation in such areas. Though the strict preservation of such extensive areas has clearly benefitted the conservation of biological diversity by preserving large areas of natural habitat for wilderness-dependent species, certain inherent aspects of the system also limit its utility for that purpose. In particular, areas not qualified to be considered for wilderness designation, because of their small size or because of roads or other evidence of human presence, may nonetheless be vitally important for their biological resources. The preservation of such resources may also require the sort of intensive management efforts that are not permitted within designated wilderness areas (Schoenfeld & Hendee). A variety of innovative administrative approaches to address this broader conservation need have been developed in recent decades.

As long ago as 1927, the Forest Service established the first of what was to become an extensive, informally connected system of "research natural areas". In the following decade the Park Service established some 28 "research reserves" in ten national parks. Later, the Park Service was to take the lead in advocating a coordinated effort among all federal land managing agencies to inventory areas under their jurisdiction and specially protect those having either representative or unique characteristics of value for research or educational purposes. The Park Service effort got a boost when, in a 1965 "Special Message to the Congress on Conservation and Restoration of Natural Beauty", President Johnson directed that a study be undertaken "to recommend the best way in which the Federal government may direct efforts to advancing our scientific understanding of natural plant and animal communities and their interaction with man and his activities." The following year the Federal Committee on Research Natural Areas was formed.

The Federal Committee on Research Natural Areas was an informal committee comprised principally of representatives of federal land managing agencies. It was entirely a creature of administrative invention; it had no statutory mandate, no appropriated funds, and no staff. Its purpose was to encourage the development of a system of specially designated and protected areas of value for research and education. Initially, the intention was to insure that natural ecological processes would be allowed to govern in such areas. Later, the system evolved to include areas where experimental manipulative management was to be carried out. The name of the committee evolved as well, becoming in 1974 The Federal Committee on Ecological Reserves. That year the committee formally adopted a charter and, by publishing it in the Federal Register, announced generally its objectives, which included expanding the existing system to include additional federal lands as well as state, local and private lands, and developing guidelines and criteria for management of the areas in the system. In 1977, the Committee published a comprehensive directory of federal research natural areas, which included nearly 400 areas totalling more than four million acres. Though still other areas were added later, within two years the committee was moribund. Though it has not been formally disbanded, the committee last met in December 1979. The special areas it helped establish continue, however.

The decline of the committee is probably attributable directly to its lack of staff and funding. Of no less significance, however, is the fact that participating agencies were ultimately bound by the varying legislative authorities that governed their respective agency missions. The committee, as an informal, ad hoc entity, could neither redirect the budgets nor reformulate the policies of its member agencies.

For one of the key participants in the committee, the Bureau of Land Management, enactment of new organic legislative authority in 1976 gave it explicit authority to do what the interagency committee had strived to do. The Federal Land Policy and Management Act required BLM to prepare and maintain a comprehensive inventory of the lands under its jurisdiction and the resources thereon. In so doing, BLM was to give "priority to areas of critical environmental concern", which were areas "where special management attention is required . . . to protect . . . important . . . fish and wildlife resources or other natural systems or processes," among other things. 43 U.S.C. 1702(a), 1711(a).

Areas of critical environmental concern, or "ACECs", are designated at the land use planning stage of FLPMA (43 U.S.C. 1712). BLM is required to draw up management plans, with public notice and comment, for each of the "resource areas" into which the agency has divided the lands within its jurisdiction. In these resource areas, or in the districts that are the subunits composing the resource areas, areas that are deemed to require some sort of special care can be designated ACECs. An area can be nominated for designation by anyone, and the Nature Conservancy has been particularly active in these nominations, using its "heritage inventories" as a basis for the suggestions it makes. Once the management plans are drawn up, they must be approved by the BLM state director.

Particular ACECs ideally have their own management plans, written to spell out in more detail the broad purposes of the resource area plan. These more focused plans are referred to as "activity plans." Very few ACECs have activity plans yet in place. BLM has not yet issued guidelines for field staff in the actual management of ACECs. This may not be surprising because "management" appears generally to be confined to fencing, posting signs, and patrolling the areas. In any case, most attention in the bureaucracy has been focused on the process of designating ACECs, by far the most controversial issue surrounding them. This process is detailed in the BLM manual at section 1617.8 _et seq_.

There has been some confusion over the distinction between the pre-existing "research natural areas" (RNAs) and ACECs. Indeed, RNAs are only one of a group of land designations pre-dating FLPMA; others include "outstanding natural areas" (ONAs), "primitive areas," and "recreational lands." As a group they are known as the "2070 designations," referring to the section of 43 C.F.R. under which they were designated. Their statutory authority was the Classification and Multiple Use Act of 1964, a statute superceded by FLPMA. Hence the 2070 designations are of unclear statutory authority. How to distinguish an RNA from an ACEC befuddled BLM regional staff in the years immediately following FLPMA, and to some extent this persists today. In fact, RNAs are enough like ACECs that of the approximately 64 RNAs that BLM administers, about half have been designated ACECs as well.

As with other parts of the BLM's budget, the general scarcity of funds for the ACEC program has been a large obstacle to proper management. There is, first of all, the budget reduction commencing in 1981. With the cuts and inflation, wildlife personnel have suffered a 20% reduction. BLM state offices are down from two or three wildlife biologists to one. BLM has tried unsuccessfully to keep constant the number of field biologists, but the ranks of plant and fish specialists have been thinned drastically. There remain only eight BLM botanists and twenty-seven fish biologists in the whole country; so there have been few new plans for areas concerned with plant and fish conservation. Wildlife ACECs have the added handicap of competing for funds with all of BLM's other wildlife programs.

Given its funding and manpower shortages, BLM appears to have had some success cooperating with private organizations in performing its statutory tasks. The Nature Conservancy is first among these, helping out especially in nominating likely areas for designation. The American Speleological Society has also contributed by mapping caves on BLM land and improving trails in them. The Nature Conservancy's general strategy has been to work locally to set in place a de facto system of ACECs when BLM has been reluctant to designate them formally. The Conservancy has also negotiated a number of cooperative agreements with BLM concerning ACECs. One of its methods has been to purchase a tract of land, negotiate with the agency for its management along certain guidelines, and then exchange the tract for a parcel of lesser BLM land that it sells to recoup its costs. Where development is occurring around a given site, the Nature

Conservancy may purchase the land, then agree to sell it to the developers on the condition that they create a preserve with a certain portion of it. This took place, for example, in the Coachella area in California. Other cooperative agreements between Nature Conservancy and BLM entail joint management of tracts. Once basic principles have been agreed upon, BLM will provide the expertise and Nature Conservancy the volunteer labor. In California, the Nature Conservancy has negotiated these agreements for three ACECs, one of which is an isolated tract, not adjacent to any Nature Conservancy land. BLM has also enlisted the help of other private organizations. In one district an ORV group helps BLM patrol, and in others archeological groups have helped support the agency's efforts.

The confusing system of protected land area designations of BLM ought to be made more uniform. This is the subject of a draft policy statement now circulating within BLM, which would draw all the old designations under the ACEC umbrella, making them all ACECs but with varying degrees of protection. The old names would be retained, but in this form: Such-and-such Research Natural Area, an Area of Critical Environmental Concern. This streamlining would serve to make uniform the designation of areas from state to state and resource area to resource area. In addition, since ACEcs are designated through the FLPMA land use planning process, it would require managers to set forth specific prescriptions, discouraging them from later downgrading the area's designation. This might mean fewer ACECs, but those designated would be more likely to receive better management.

### 3. Private Lands

In addition to the various systems of natural area protection on federal lands, there are a variety of federal laws and programs designed to encourage the conservation of biological resources on private lands. One of the most interesting of these is the National Natural Landmarks Program, established in 1962 and administered by the National Park Service. It is similar to the National Historic Landmarks Program; indeed, its claimed statutory basis is a somewhat generous reading of the general policy declaration of the Historic Sites Act of 1935, 16 U.S.C. 461 et seq. Its purpose is to "identify and encourage the preservation of the full range of ecological and geological features that are nationally significant examples of the Nation's natural heritage." 49 Fed. Reg. 4605 (Feb. 7, 1984).

Currently, more than 550 areas are included in the National Natural Landmarks program. About half of these are owned or administered in whole or in part by private interests. The designation of a landmark does not change its ownership status, nor does it result in any regulatory consequences. Owners of designated areas receive recognition in the form of a plaque and consultative assistance in protecting the site. The underlying premise of the program is that many private landowners, when made aware of the ecological or other significance of the lands they own, will cooperate voluntarily in the protection of those lands. Indeed, when an area is designated as a National Natural Landmark, the owner is encouraged to agree to preserve the natural values of the site. That agreement can be terminated unilaterally, however, and does not bind successors in interest. Thus, while the program offers a means of encouraging private initiative in the conservation of areas of major biological significance, its potential is limited by the fact of its purely voluntary nature. Indeed, since 1970 Congress has required the Park Service to report annually to it on designated landmarks that are threatened or damaged. The Service's must recent report identifies 21 such sites, most of which are threatened by on-site activities. The National Landmark Program could, if implemented more aggressively, be a very important means of advancing the conservation of biological diversity on private lands. Currently, however, its considerable potential is being largely unrealized.

An older and more diverse set of programs to encourage the conservation of wildlife on private lands is administered by the Agriculture Department. These include the Water Bank Program, in which farmers in the Northern Plains are paid in return for agreeing not to drain wetlands on their properties, the Agricultural Conservation Program, in which the federal government provides loans or partial funding for various activities to promote conservation of wildlife and other resources, and the Forestry Incentive Program, which provides assistance to private landowners for improved production of timber and related benefits, including wildlife.

Traditionally, the Agriculture Department programs have emphasized the recreational benefits of wildlife, primarily hunting and fishing. As a result, increasing the supply of game animals, rather than fostering the conservation of biological diversity itself, has been the principal wildlife concern of these programs. However, considerable biological diversity benefits could also be derived from these programs, such as those that could be derived from improved private forestry practices aimed at aiding the endangered red-cockaded woodpecker, to give but one example.

### Other Approaches

Yet another approach to the problem of conserving biological diversity is reflected in the many federal statutes that require consideration of wildlife or other environmental impacts when agencies authorize or undertake activities that may affect those resources. The most general of these is the National Environmental Policy Act, which applies to all federal agencies and requires them to give full consideration to the environmental impacts of any major action that significantly affects the quality of the human environment. The means of giving that full consideration is through the preparation of a detailed "environmental impact statement." Having prepared such a statement, however, a federal agency is free to carry out its proposed action, no matter how detrimental its environmental impacts may be. In short, NEPA imposes important procedural obligations that ensure the full disclosure of the environmental effects of federal agency action, but it imposes no substantive environmental standards that those actions must meet. Assessing the contribution of NEPA to the conservation of biological diversity in a rigorous, quantitative way is probably impossible. Clearly, its full disclosure requirements have served to reduce or eliminate the adverse effects of many planned projects; equally clearly, however, there is much environmental destruction by federal agecies that NEPA has not prevented.

Similar duties to "consider" the impacts of proposed federal actions are found, in one form or another, in a wide variety of other, more specific, federal laws. The Fish and Wildlife Coordination Act, for example, imposes such a duty on federal agencies undertaking water resource development projects or authorizing private persons to fill or modify aquatic areas. Because it lacks substantive teeth, the Coordination Act has been of limited effectiveness. More effective has been the duty to route highways around parks and wildlife refuges unless there is no "feasible and prudent alternative" to doing so, imposed by Section 4(f) of the Department of Transportation Act. The Federal Power Act, in yet another variation on the theme, requires that certain wildlife conservation recommendations of the U.S. Fish and Wildlife Service be incorporated in hydroelectric licenses issued by the Federal Energy Regulatory Commission. Because these recommendations must be included in the licenses, the Power Act gives a firmer legal basis for conserving biological resources than does the Fish and Wildlife Coordination Act.

Because of NEPA's general applicability to all federal agencies, no agency is free to ignore completely the environmental — and biological diversity — consequences of its programs. To the extent that certain federal programs can be said to be hindering the conservation of biological diversity, therefore, that result is often a consequence of the lack of more specific substantive standards in the applicable legislation and the agency's unwillingness to accommodate fully environmental goals in carrying out its primary mission.

Some of the key federal laws or programs that most adversely affect the conservation of biological diversity do so as a result of direct or indirect subsidies to development in ecologically significant areas. The rapid conversion of bottomland hardwood forests in the lower Misissippi River valley to soybean and other agricultural uses is the result, at least in part, of federal crop subsidy programs. So too has the underpricing of water and energy from federal reservoirs, particularly those of the Bureau of Reclamation, encouraged the building of still more reservoirs and the destruction of many miles of riverine habitat to supply agricultural and industrial activities that would not be feasible without such subsidies. Other federal subsidy programs have encouraged overinvestment in port and waterway transportation systems, commercial fishing fleets, and other enterprises, often with significant adverse environmental effects.

Reducing the adverse environmental effects of federal subsidy programs does not necessarily require the elimination of subsidies. For example, if existing beneficiaries of subsidy programs are able to market their "rights" to participate in those programs, the distorting effects -- and the adverse environmental consequences -- of the programs can be eliminated. The recent agreement between the Metropolitan Water District of Los Angles and the Imperial Valley Irrigation District illustrates the potential for creative, market-based solutions to major environmental problems. Under that agreement, Los Angeles will invest in irrigation efficiency improvements for the Imperial Valley farmers and in return will receive the water that such investments conserve. As a result, Los Angeles can obtain sufficient new water for several decades of anticipated growth without the need to build new storage facilities in pristine rivers in northern California or already stressed rivers elsewhere. Facilitating such transfers of subsidy benefits should be a high congressional priority both because it can produce major environmental benefits and because it does not entail the politically difficult task of removing subsidies from existing beneficiaries

## Conclusion

The various species, natural areas, and indirect conservation laws of the federal government provide an elaborate framework upon which a concerted biological diversity conservation effort could be built. The one law that clearly has the conservation of biological diversity as its central aim, the Endangered Species Act, has thus far lacked the resources to carry out its fundamental purpose successfully. However, the work done under that Act, and in particular the identification of a large number of plants and animals in the United States that are legally unprotected but nevertheless in peril of extinction, provides an opportunity for other agencies administering other programs to coordinate those other programs with the Endangered Species Program in ways that would substantially aid the conservation of biological diversity. For example, a concerted effort to seek National Natural Landmark designations for key habitat areas of the candidate plants and animals would serve to enlist many private property owners in the biological diversity conservation effort. A similar concerted effort to designate such areas on federal lands as Areas of Critical Environmental Concern, Research Natural Areas, or similar designations, would greatly enhance the cause of biological diversity in the third of the nation that is in federal ownership. Expanding the traditional limited focus of Agriculture Department wildlife conservation programs to encompass explicitly biological diversity concerns would hold out the potential for similar benefits in the vast areas of the nation devoted to agriculture.

In most cases, no new legislation would be needed to make the changes in emphasis suggested here. Rather, some clear indication of congressional desire that the potential of these existing programs for aiding the conservation of biological diversity not go unrealized would suffice. The first step that Congress could take to give this signal is to request of these various agencies administering these various programs detailed reports of how those programs are being implemented with respect to the goal of conserving biological diversity, and in particular to identify what measures are being undertaken to use existing program authority to aid legally unprotected but biologically vulnerable candidate species.

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# Ecological Restoration As A Strategy for Conserving Biological Diversity

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# ECOLOGICAL RESTORATION AS A STRATEGY FOR CONSERVING BIOLOGICAL DIVERSITY

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**REVISED DRAFT**

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# ABSTRACT

Though the restoration of disturbed ecosystems has so far played a relatively modest role in the effort to conserve biological diversity, there are reasons to suspect that its role will increase and that its contribution to the maintenance of diversity will ultimately prove crucial as techniques are further refined and as pristine areas for preservation become scarcer and more expensive.

For one thing, the destruction of pristine ecosystems will continue in the future, often at an accelerated rate and as the result of forces over which conservationists have little or no control.

For another, techniques for the reclamation and even the restoration of disturbed ecosystems have improved rapidly in recent years. As a result, it is now possible to restore a number of North American communities with some confidence. However, it should be noted that many current efforts to return degraded lands to productive use, like attempts to reclaim land disturbed by mining, try only for rehabilitation to a socially acceptable condition and fall considerably short of actually restoring a native ecological community.

Possible uses for restoration in the conservation of biodiversity include not only the creation of habitat on derelict sites, but also techniques for enlarging and redesigning existing preserves. Restoration may even make it possible to move preserves entirely in response to long-term events, such as changes in climate. Restoration in the form of reintroduction of single species to preexisting or restored habitat is also a critical link in programs to conserve species *ex situ* in the expectation of eventually returning them to the wild.

Opportunities for employing restoration to increase diversity are numerous and include landscaping, the management of parks and preserves, management of utility corridors and transportation rights-of-way, as well as restoration of areas disturbed by influences such as pollution, mining, construction, or agriculture.

Measures that might encourage restoration-oriented reclamation include support of appropriate research and improvement of communication both within the discipline itself and between restorationists and the public. In general, this field is neither clearly defined nor well documented, and a systematic survey of it would be an important step toward evaluating its potential as a form of environmental technology.

## INTRODUCTION

Plans for conserving biodiversity have tended to emphasize in situ preservation of naturally occurring communities. While vital, such efforts at in situ conservation cannot alone preserve all species or communities, given such threats as increasing development pressure, changing climate, and growing human populations. Fortunately, there are additional tools conservationists can use. One strategy, which has generally been overlooked, but which holds great promise, is that of restoration. In restoration, the goal is to return degraded biological communities to their original state with human help. Though such efforts have generally played a minor role in conservation efforts aimed at conserving diversity, it seems likely that this role will increase as pristine habitat becomes scarcer and as opportunities to "recycle" ecosystems disturbed by human activities and unusual natural events become more common.

In this paper we will make a distinction between two terms that are poorly distinguished in the literature: By "restoration" we mean the recreation of entire communities of organisms, closely modelled on those occurring naturally, while we will use the term "reclamation" in a more general sense to refer to any deliberate attempt to return a damaged ecosystem to some kind of productive use or socially acceptable condition short of restoration. This may, for example, involve revegetation of a mined area with exotic vegetation. The distinction between restoration and reclamation is not a sharp one however. Reclamation grades into restoration as more and more naturally occurring species are used, and the development of restoration technologies depends heavily on past reclamation experience.

Several other definitional points should be made. The first is that no sharp distinction can be made between the reintroduction of single species and restoration of species-complexes, although restoration clearly becomes more complex as more species are involved. Second, restoration grades into management, and many common management techniques such as control of exotics and periodic burning are used in both the maintenance of extant communities and in the restoration of degraded ones.

Finally, as for the term biodiversity itself, we will accept as a standard the diversity characteristic of the natural communities that provide the models for restoration. If, as is usually the case, the species diversity of a site has been decreased as a result of disturbance, then restoration will result in an increase in diversity. In some cases, however, where the diversity of the original community is low, or where restoration entails mainly the removal of exotics, restoration may actually lead to a decrease in the number of species present. Even in these cases, however, restoration will usually result in an increase in what might be called the "native" diversity of the site -- that is, an increase in the number of species present that were also present in the pre-disturbance community.

This report represents a general overview of restoration and the role it might play in conserving biological diversity. We will explore the state of the art for restoring various kinds of communities and will consider the techniques involved, their cost and effectiveness, and the length of time they take to significantly contribute to diversity. We note at the outset, however, that this is a formidable task, given both the complexity of the field and the present state of communication in it. For both of these reasons, this must be regarded only as a preliminary effort.

## RESTORATION OF MAJOR NORTH AMERICAN ECOLOGICAL COMMUNITIES

Alterations in the landscape have occurred almost everywhere on the North American continent since the time of European settlement, and it might be supposed that wherever such alterations have occurred restoration might contribute to the recovery and conservation of biodiversity.

In the following summaries we have attempted to consider those factors that are of major importance in evaluating the promise of restoration as a conservation strategy. These factors include the natural diversity of the community itself, the extent to which it has been reduced or changed in character by human activities since the time of European settlement, variations in the difficulty of restoring the various communities, differences in the amount of effort that has been devoted to developing various restoration techniques, and also the motives for doing restoration. Motives may vary considerably and in some cases are only indirectly related either to genuine restoration or to the conservation of biodiversity.

Given limitations on time and space, and also given the extremely scattered and uneven nature of the literature in this field, it has not been possible for us to deal with all community types. Instead, we have chosen communities that are of special interest, either because they have been dramatically reduced or altered since the time of settlement or because they have attracted the special attention of restorationists interested in diversity. While the state of the art varies from community to community, it should be kept in mind that in each case questions remain and no community can always be restored with complete confidence.

**Prairies of the Central United States**

The prairies and savannas of central North America have a special significance from the point of view of biodiversity because, while they were once among the most extensive and floristically rich communities native to the continent, they are now among those most reduced by post-settlement activities such as agriculture, development, grazing, and the introduction of exotic species.

Partly for this reason, and also partly because prairies lend themselves to restoration on a modest scale, in reasonable lengths of time, and by more or less standard agricultural techniques, restorationists have long taken a special interest in prairies. Moreover, because prairies are often of relatively little direct economic value in themselves, efforts to restore prairies have frequently been true restoration efforts, motivated by ecological concerns such as diversity or community authenticity. Such efforts contrast with reclamation attempts where, for example, the goal might be reforestation with any tree species that can efficiently produce a salable crop. As a result, prairie restoration has been a prototype for the development and use of restoration techniques.

Midwestern prairies and savannas have been profoundly influenced by post-settlement human activities. By 1967, for example, the area covered by bluestem prairie had been reduced by 88 percent since the time of European settlement, (Klopatek, Olson, Everson and Jones, 1979) and much of what survives is in a more or less degraded condition. Since prairies are relatively rich in species, many of which are found only in prairies, their loss represents a major reduction in the biotic diversity of the North American continent. Thus, the successful restoration of even relatively small examples of these communities may be a major contribution to the restoration of the native biodiversity of an area.

Comprehensive data on the extent of existing restored prairies are not available, but we do know that the percentage of prairie acreage that is the result of deliberate restoration efforts is probably still small. Of 22 prairies totalling 214 ha listed as having preserved status in Michigan, for example, only five, totalling 7 ha are restored (Chapman and Pleznac, 1981). Nevertheless, it is significant that two of the largest prairies in Wisconsin, another state on the northern border of the prairie province, are restored communities.

While prairie restorationists frequently draw attention to the complexity and uncertainty of their work (Schramm, 1978; Morrison, 1981; Powers, 1984), a number of successful techniques have been devised for the restoration of prairies under various ecological and economic conditions (Schramm, 1970, 1978; Rock, 1976; Warkins and Howell, 1983; Waltham and Gilbert, 1978; Morrison, 1981; Holtz and Howell, 1983; Smith, 1980). In general, prairie restoration techniques borrow extensively from agriculture, especially from horticulture and, for larger-scale work, from agronomic techniques in use in the prairie region. A recent summary of techniques for restoring prairies of between two and 40 ha recommends plowing, followed by discing at intervals of a year to reduce weeds, followed by seeding with a mixture of prairie species. Hydroseeding and mulching with a wood fiber slurry may be desirable on steep, erodable slopes such as may be encountered on rights-of-way (Morrison, 1981).

Since biodiversity is often a major consideration in prairie restoration, and since restorationists have frequently experienced difficulty adding species to established prairies, they frequently advise starting with as complete a mixture of species as possible. In contrast, attempts to develop more efficient methods of restoring the large (240 ha) prairie at Fermi National Laboratory near Chicago have led to the introduction of species in a sequence of "waves" roughly corresponding to the waves of succession that might be expected to take place on a prairie developing naturally on disturbed land (Betz, 1985). Although a variety of techniques, including careful timing of planting, mowing, burning, and the use of nurse crops have been used to control weeds during early stages of prairie development, weed growth remains a serious problem limiting the success of prairie restoration.

In addition to restoration essentially from bare ground, restorationists frequently undertake restoration or ecological upgrading of existing prairie remnants degraded by influences such as mowing, grazing, or protection from fire (Betz, 1985; Holtz and Howell, 1983). Work of this kind has led to the realization that former prairies occupied by forest following fire suppression may retain a cryptic prairie vegetation for long periods, and that this may recover rapidly if trees are removed and appropriate management techniques initiated (Curtis, 1959). This technique has proved highly successful even after as much as

70 years, suggesting the possibility of bringing back many acres of prairie and recovering numerous local ecotypes of prairie species that might otherwise have been regarded as extirpated (Curtis, 1959, p. 304; Holtz and Howell, 1983).

As always in the area of ecological restoration, the question of the authenticity of the resulting community is crucial and is widely discussed. So far as we know, no one claims to have restored a fully authentic prairie, (e.g., Sperry, 1983), but in general, it has been possible to create prairies with lists of plant species closely resembling those of the model community and at least some native animals.

It might be noted here that, in the case of prairies, as in restoration generally, restoration efforts usually focus on revegetation on the assumption that if the plant cover is restored, the animal life will reappear naturally. This, of course, is true only if there are adequate sources of animal colonists and if the plant cover chosen provides the resources necessary for the animals. These assumptions may hold true for many animals, like some birds, that are highly mobile and not highly specialized. However, other species may find it difficult to recolonize restored habitat, and more attention to the reintroduction of such species might significantly increase the biodiversity value of some restoration efforts.

In sum, the comments of numerous experienced restorationists suggest that prairie restoration techniques, while far from adequately refined, do make it possible to create prairies with a high level of native diversity, at least as far as plants are concerned. Indeed, the restored prairies at the University of Wisconsin Arboretum include more than 400 species of plants alone, most (though not all) of which are prairie natives, and these prairies have been described as being among the richest in species of any plant community in Wisconsin (Cottam, 1979). Morrison (1981) states that "There is now sufficient experience and documentation of prairie restoration work to suggest that this alternative (i.e., restoration) is a viable one to pursue, especially on large land areas."

At the same time, prairie restoration is a demanding, time-consuming process, and there is difficulty in obtaining seed and plant stock, especially of local ecotypes, which often results in the use of cheaper non-local seed, often from relatively few species (Matthiae, 1985). In general, such factors have limited the scale of prairie restoration efforts. For example, the Fermilab prairie, by far the largest tallgrass prairie restoration effort yet undertaken, now includes only 180 ha after 11 years of effort. Work is now proceeding there at a rate of about 40 ha per year.

There are numerous ways in which the goal of prairie conservation through restoration may be married with other goals of social importance. Thus, prairies are being restored on rights-of-way, along utility corridors, and in parks and nature preserves (Riskind and Davis, 1975). They are being created as wildlife habitats (Camp, 1985; Woehler, 1985), to provide warm-season pastures (Maddy, 1970; Anonymous, 1983), and on land disturbed by mining (Morrison, 1982), construction, or dredging (Green, Sifuentes and Martin, 1981). Others are created for landscaping (Diekelman and Schuster, 1982) and educational purposes. Another important "use" for restored prairies is restoration of soils degraded by agriculture, and indeed recent evidence suggests that prairie may restore the crumb structure of certain soils more effectively than any known kind of vegetation (Miller and Jastrow, 1985).

While it is widely accepted that prairies are less costly than many more traditional forms of landscaping, little information on the cost of prairie restoration is apparently available. Up to now, much of the effort that has gone into restoration of the highest quality prairies has tended to depend heavily on volunteer effort (e.g., Betz, 1985).

## Western Grassland, Shrub-grassland, and Savanna

As in the central United States, western grasslands have experienced exceptional losses since settlement times, with three vegetation types experiencing destruction of 75 percent or more of their land area (Klopatek et al, 1979). Factors contributing to the destruction of these communities have included conversion for agriculture, grazing, invasion by exotic species, and deliberate reseeding of depleted or shrub-dominated rangelands with monocultures of exotic grasses.

Many land reclamation projects have been undertaken in the West in recent years. While few qualify as restoration, some do emphasize the reintroduction of native species -- under current regulations even reclamation requires the use of native plants and has led to the development of techniques that will be valuable for restoration projects. Mine reclamationists in the West, for example, have made extensive use of the technique of transplanting native species into reclamation sites from areas about to undergo stripping (Brown and Hallman 1984). This lends an aura of restoration to mine reclamation, since these species are often replaced in patches and form islands of native flora and soil microorganisms. This technique could be used in other development projects where certain plant species need to be saved, but caution must be exercised to ensure that relocated plants are placed in suitable sites.

Although reclamationists may have taken steps toward developing technologies for restoration, problems remain. One largely technical problem limiting restoration efforts is that much revegetation equipment is designed to seed or plant in rows. To avoid this, seed may be broadcast or seedlings may be planted by hand in more natural patterns. Hand planting, however, may be expensive, and broadcasting of seed is often less successful than drilling in arid regions.

Another problem is lack of information about native species and the dynamics of native communities. For instance, there are limited data on the seed germination and seedling phsiology of many western species, and mixtures of warm- and cool-season grasses are still difficult to establish because the physiological responses of the various species differ widely and are poorly understood (Brown and Hallman, 1984).

As in prairie areas farther east, supplies of seeds and plants may also be a problem. Although western nurseries and the Soil Conservation Services Plant Materials Centers have responded to the demand for more native species, many species are still not commercially available. Often, only specialized ecotypes or registered cultivars with limited value for restoration are available. Use of local seed is not required by law, and government agencies and private industry are reluctant to supply local varieties because they are relatively expensive.

There is also uncertainty about the best way of developing a diverse community of native species on a disturbed site. Reclamationists frequently say that succession and colonization of reclaimed sites by native species will eventually result in a richer, more natural community. However, revegetated sites in Wyoming that were up to 10 years old have not experienced appreciable colonization by nonplanted native species (Parmenter et al., 1985). This may be due to the initial conditions at the sites where the seeded vegetation may be so dense that there is little opportunity for colonization by non-weedy colonizers. High density seeding is done to assure quick stabilization and adequate seed to germinate in dry years, but this conflicts with the goal of increased richness. A number of different cultural practices, including a more diverse seed mix and interplanting of species that do not establish well by seed could be employed to increase richness.

The scarcity of undisturbed relict areas to serve as reference areas for judging the success of restoration may be the most important limitation to restoration (MacMahon, 1983). In areas like the Palouse prairie, the California grasslands and savanna, the Great Basin and many western shrublands which have been dramatically altered by grazing, agriculture, and other disturbances, there may be no quantitative record of the original vegetation. In such cases, of course, a quantitative comparison of reference and restored areas is not possible, and the restorationist may have to be content with a rough qualitative match.

Cost is also an important consideration, since, other considerations being equal, restoration is likely to be more expensive than other forms of reclamation. A minimal cost of wildlife habitat improvement in Utah, for example, is $125/ha (Fairchild, 1985). This includes seedbed preparation, a seed mixture of 12 species, and broadcast seeding. The average cost of seeding reclaimed surface mines in seven western states is $620/ha (1977 dollars) (NAS, 1981). But these efforts rarely represent restoration, which is likely to cost several times as much; one study in Utah suggests a cost of from $500 to $2,000/ha (Fairchild, 1985) depending on whether containerized or bare-root stock was used. Hand planting to simulate natural vegetation patterns would increase costs even more. The cost of planting shrubs, grasses and forbs in a experimental restoration project in Wyoming was about $8,500/ha (MacMahan, 1985).

Although seeding cost are considerable, at least on mine sites the preliminary earth-moving costs far more, so that total reclamation costs may easily be $10,000/ha, whether a site is restored or merely revegetated.

A final factor limiting the use of restoration is the nature of regulations pertaining to reclamation of disturbed lands. Mining companies are generally not required by law to plant more than five to ten species. In addition, grading techniques specified by law tend to reduce micro- and meso-scale diversity. After reclamation, uniform slopes with uniform topsoil depths are usually seeded with uniform seed mixtures. Some western state regulatory agencies, however, are beginning to require increased landscape diversity by requiring variations in contours, topsoil depth and seeding mixture.

## Forests

The status of forest restoration as a strategy for conserving biodiversity contrasts in a number of ways with that of prairie restoration. For one thing, smaller fractions of most forest types have been lost as a result of human activities since the time of settlement, and even where forests have been affected by human activities, they have often not been destroyed outright, but only disturbed by activities such as logging, which permit at least some form of natural recovery. For another, tree-planting may well be one of the oldest and most frequently used techniques for the reclamation of disturbed land. As a result, there is a large and relatively well organized literature on various forms of reforestation (e.g., USDA-Forest Service, 1985a). Moreover, certain forms of sustained-yield forestry tend to preserve or increase diversity and deserve to be regarded as a form of restoration.

At the same time, far less work aimed specifically at increasing native diversity has been done with forests than with prairies or with certain other, smaller-featured communities such as wetlands. One reason is that forests, unlike prairies generally, often have a high economic value which might be best realized in a highly managed "undiverse" forest. Another reason is that the length of time required to restore a forest of long-lived trees may make such restoration less tractable. Forests may also be more difficult to restore because they are rich in species and structural complexity -- the temperate forests of North America may include a thousand species of vascular plants alone (Ashby, 1986).

Thus, in general, and with occasional exceptions (e.g., Brown, Maddox and Splittstoesser, 1983), most of the research in this area appears to be oriented toward the performance and use of one or a few species at a time. Asked for examples of diversity-oriented work on forest restoration, W. Clark Ashby wrote, "I still have not thought of a forest study geared to increase diversity," (Ashby, 1985). Nevertheless, there is growing interest in the more restoration-oriented forms of forest establishment, an interest that is motivated partly by a desire to create wildlife habitat (e.g., Rafaill and Vogel, 1978). Evidence for this interest is the formation of the Fish and Wildlife Relationship to Mining Committee, created within the American Fisheries Society in 1982 to encourage the creation of "wildlife habitat of high ecological quality" (Starnes, 1984). Development of wildlife habitat may or may not imply restoration of a fully authentic native community, but it does imply a higher level of biotic diversity than is required for such things as site stabilization or timber production (Vogel, 1977), and it often represents at least a step toward genuine restoration (Starnes, 1985; Noss, 1985a).

It is also true that forests or plantations established on disturbed sites frequently do achieve a high degree of diversity as a result of invasion by "volunteer" species. This has been documented on reclaimed mine sites in the lower Midwest, for example, (Kolar and Ashby, 1978; Ashby, Rogers and Kolar, 1980; Applegate, 1985), and in Ohio (Larson, 1984).

So far as we are aware, very few genuine forest restorations have been undertaken and even fewer have been monitored over any length of time. Perhaps some of the oldest and best-studied are at the University of Wisconsin-Madison Arboretum, where restoration of a variety of pine, boreal, and maple forest types has been underway since the 1930's. Here, success has varied widely, reflecting differences in, for example, the suitability of the soils or the local climate for the forest types attempted (Wildman-Evans and Albright, 1984; Cottam, 1979). Other more recent examples of forest restoration work include restoration of 16 forest types native to New York state near Brockport, NY (Hellman, 1984) and the large-scale restoration of redwood forest on 14,500 ha of logged land added to Redwood National Park in 1978 (Belous, 1984).

### Coastal Salt Marshes and Segrass Communities

Coastal marshes are found in the intertidal zones of moderate to low-energy shorelines along tidal rivers, estuaries, and bays where the plants are periodically flooded by salt or brackish water (Woodhouse and Knutson, 1982). They are not only unusually productive and valuable ecosystems (Woodhouse and Knutson, 1982), but they are also extremely vulnerable to development, and losses of these communities have been dramatic. Since 1954 1/2 million ha have been lost nationwide (Gosselink and Baumann, 1980). Although the East coast still has one million ha of wetlands, only some 41,500 ha remain on the U.S. Pacific coast (see in Knutson and Woodhouse, 1982). Although Section 404 of the Clean Water Act (U.S.C., 1981) has slowed direct destruction (Army Corps, 1982), many problems remain because mitigation measures often prove inadequate and because losses due to various types of pollution continue (Zieman, 1985; Zedler, 1985).

Because the plant composition of salt marshes is relatively simple it might be expected that they would be especially amenable to restoration attempts. Indeed, the last decade has seen a large increase in restoration efforts involving coastal wetlands, often the result of mitigation required by regulatory bodies. Such restoration efforts have been successful both at replacing destroyed vegetation on the original sites and at planting marsh vegetation where none existed before (e.g., in Jordan, 1983).

Although there is a variety of saltwater wetlands that differ from each other in species composition and general ecology, a number of generalizations can be made about restoration of these communities.

A first generalization is that the field of marsh restoration is still young. Although it has been amply demonstrated that coastal marshes can be created under a variety of circumstances (see Woodhouse and Knutson, 1982 for a comprehensive survey of U.S. plantings), much is unknown about planting techniques and about the ecological requirements of the plants. For example, establishment of cordgrass, the species most frequently used on the East Coast, is still restricted to areas of low wave energy because of washout problems, although the Army Corps of Engineers Waterways Experiment Station at Vicksburg, Mississippi is experimenting with methods of establishment in areas of higher wave energy (Allen, 1985). According to Lewis (1982) restoration of salt water marshes would benefit by the selection of especially robust strains of commonly used plants, and by the development of better sources of plants and seeds. At the same time, work has progressed so well with some species that private contractors are now able to guarantee as much as 80 percent survival of some salt marsh plants under favorable conditions (Lawrence, 1985).

A second generalization is that, as in the case of much reforestation work, plantings have generally been of only one, or at most a few species, and there has been little interest in inoculating sites with rarer or more difficult plants. One reason for this is that many of these communities are naturally low in floristic diversity, some of them approaching monocultures in composition. Other reasons include the facts that it may be cost effective to standardize propagation and planting by using a single species, that it is easiest to use those few species for which efficient propagation techniques have already been developed, and that certain, commonly used species may be the best for particular uses, such as beach stabilization. Fortunately for diversity, many species are able to invade a newly created marsh as volunteers.

A third generalization is that there is little planning for the transplantation of animal species from existing marshes into new ones, and many animals may not be able to recolonize by themselves. For example, in the Pacific coast marshes, which are ecologically discrete entities typically located only in the mouths of bays and lagoons, populations of both animals and plants may be in relative isolation from each other, so that natural reestablishment of populations following local extinctions is often unlikely. Clearly, replacement of a developed salt marsh with a new one in a different location would do little good to the localized animal (or plant) species that cannot reach the new one.

A fourth generalization is that there is a great deal of variability in the degree to which restorationists are concerned with the genetic provenance of their source materials. There seems to be a growing understanding both that it is important to prevent pollution of wild strains with introduced strains from other areas, and that the best results often can be obtained by planting local ecotypes (Zedler, 1985). Thus, using plants from local sources has often increased planting survival rates and prevented genetic swamping of local types. However, this is only a dawning perception, and the "if it will grow there, put it there" philosophy is still prevalent.

Fifth, much marsh restoration has been task oriented, that is, motivated by some purpose other than that of preserving the genetic constituents of the marsh for their own sake. In many cases, marsh is used to stabilize dredge spoils, to trap sand for the commencement of dune formation, and to prevent erosion of shorelines (Soots and Landin, 1978). In other cases marsh may be created as habitat for fishery or wildlife resources. But in general the focus has been on finding a plant that will do a particular job, whether it be preventing erosion or feeding fish, and the assumption has been that one plant that does the job is as good as another.

All the preceding generalizations have bearing on the principle question: How much can marsh restoration technology contribute to the preservation of biological diversity? The easy answer is that the technology is being developed to replace lost marshes with analogous marshes. It is clear however that there is a long way to go, and that much development of the field will be necessary, both technically and philosophically, before bodies regulating marsh restoration will demand more than a "facsimile," composed of hardy, easily established plants of mixed parentage or unknown provenence, volunteer plants with good colonization powers, and a fauna of opportunistic colonizers.

Thus, while numerous tidal marshes have been created in recent years, their authenticity, especially with respect to normal tidal marsh functions, has been questioned by researchers (Race 1985; Race and Christie, 1982) who underscore the need for further research involving comparisons between natural and restored marshes and refinement of restoration techniques before restoration is regarded as an environmentally acceptable procedure, especially for mitigating the destruction of natural wetlands.

**Freshwater Wetlands**

Freshwater wetlands present an example of a classical form of restoration, largely motivated by at least partly economic interests -- in this case creation of game and wildlife habitat -- that nevertheless has had important implications for the conservation of biodiversity.

Restoration of these wetlands is complicated by the fact that they are generally richer in species than tidal marshes, and are subject to more irregular variations in critical environmental factors, especially water level. Nevertheless, according to Charles Newling (1985), a wetland manager at the U.S. Army Corps of Engineers Waterways Experimental Station in Vicksburg, Mississippi, it is now possible to restore freshwater wetlands with some confidence, especially on sites where such communities originally existed.

According to Newling, the single most important factor in freshwater restoration is restoring water levels and appropriate seasonal variations in levels, and when this is done properly considerable community development typically occurs even in the absence of further active restoration efforts. Work of this kind, he notes, has also revealed the existence of rich propagule banks in soils as much as 50-100 years following loss of the original vegetation, recalling the potential of seed banks in the soils of former prairies as a potential source of ecotype diversity (Van der Valk and Davis, 1979). In other situations, as on sites newly created by deposition of dredge spoils, introduction of propagules may be critical to the restoration of native vegetation. Newling notes that restoration work on freshwater wetlands, as in tidal wetlands, has tended to focus on relatively few species and to rely on colonization by volunteers to increase diversity. He says he knows of very little research on introduction of species specifically to increase diversity.

A major problem in freshwater restoration is the tendency of these wetlands to be invaded by exotics such as Phragmites australis or purple loosestrife (Lythrum salicaria). Newling notes that under appropriate conditions Phragmites can be controlled effectively by prolonged flooding, though control measures for purple loosestrife are not available, and this species is a cause of serious concern on the part of wetland managers in many parts of the country.

The problem of invasion by exotics is closely related to another important issue in ecological restoration -- the longevity of the resulting communities. While regulations governing reclamation practices typically may only require that a restored community be present for a relatively limited period of time following disturbance, in order to play a significant role in the conservation of biodiversity restored communities ought ideally to have life expectancies comparable to those of natural communities. Unfortunately, the lifetime of a restored community may be limited if it is restored only "cosmetically," with, for example, improper hydrological functioning.

Costs of restoration, Newling says, vary widely depending on the suitability of the site for wetland. A typical range may be from $2,000-2,500/ha, but costs may be much higher, especially on upland or otherwise naturally unsuitable sites.

**Lakes and Rivers**

The United States includes 37,000 publicly owned freshwater lakes (Hanmer, 1983) and tens of thousands of kilometers of rivers and streams. While relatively few of these have actually been completed destroyed, as have large areas of terrestrial communities, human activities have altered many of them so profoundly that the biotic communities they support bear little resemblence to those of presettlement times. For example, an estimated 85 percent of the nation's lakes are degraded to some extent, and roughly a third require some form of restoration and protection (Hanmer, 1983). Similarly, rivers and streams have suffered from channelization, pollution, and sedimentation resulting from activities such as logging, agriculture, livestock grazing, and mining.

Since lakes are valued by society not only as sources of water, but also because of their value for recreation and as fish and wildlife habitat, efforts to restore -- or at least to clean up -- lakes were among the earliest restoration efforts, dating back at least to the later part of the last century. Such efforts received a major impetus, however, with passage of the Federal Clean Water Act in 1977. Among other things, this act established the Clean Lakes Program to help states develop programs to address lake problems. Under the Act, $78.6 million had been spent by 1983 on measures to diagnose and restore degraded lakes (Hanmer, 1983).

With the encouragement of the Act, numerous techniques for lake restoration have been devised and subjected to testing. Those listed in a new book on lake restoration (Cooke, Welch, Peterson and Newroth, 1986) include nutrient diversion, dilution and flushing, phosphorous inactivation by precipitation, sediment removal, seasonal water drawdowns, macrophyte harvesting, and covering of sediments to prevent macrophyte growth.

Here, as in the case of most other communities, efforts to "restore" lakes are frequently motivated more by an interest in improving the lake for human use than by a desire to reestablish a native community of plants and animals. Nevertheless, these goals are frequently compatible, and in many instances cleaning up a lake actually does involve removing exotic macrophytes and altering conditions to more closely resemble those of presettlement times.

Comments from Lake and Reservoir Restoration (Cooke, et al., 1986) convey an expert assessment of the state of the art in this area:

"An obvious finding, while not always stated explicitly in the text, is the small amount of knowledge we actually have about protection, management, and restoration of our freshwater resources. At present, our solutions seem to be based almost exclusively upon the application of chemicals or the introduction of machinery. In many cases these approaches are effective, but they are also expensive. The developing interest in biological control may greatly add to our arsenal of effective and inexpensive techniques. This avenue may become extremely important in the near future as the demand for clean water increases and the level of tax dollars available for lake restoration decreases...

"We urge our colleagues in aquatic ecology to turn their attention to these matters."

Restoration of channelized waterways, of which there are many thousands of miles, represents yet another rapidly developing field with which we cannot deal in detail here. It is our impression, however, that traditional waterway management techniques typified by channelization have begun to give way to ecologically more sensitive approaches involving careful attention to water quality and hydrology, as well as to the development of ecological communities both in the waterway and along its banks (Willeke, 1983; Gore, 1985). The first North American Riparian Conference was held in Tucson, Arizona in 1985, and included a scattering of papers on river and stream restoration (USDA Forest Services 1985b). Major river management projects oriented toward restoration are currently planned for the Des Plaines River in Illinois and the Kissimmee River in Florida (NOSS 1985b). The Kissimmee is of special interest, since it was channelized during the 1960's and is now to be in large part returned to its original, meandering bed.

## "Minor" Communities

Some types of ecological communities cover relatively small areas, yet are important constituents of total biological diversity. We will not cover them in detail here, but note that examples include beach and cliffside communities, as well as some types of prairies and wetlands, such as grassy balds, bogs, and fens. Efforts to restore examples of such communities have generally tended to be undertaken informally and on a small scale, frequently by amateurs guided by horticultural and ecological instinct. Though these small projects often more closely approach true restoration than do some larger projects, these small projects are often unreported, and there is little literature in this area. However, numerous examples may be found scattered through the eight issues of "Restoration and Management Notes" published since 1981.

## SPECIAL APPLICATIONS OF RESTORATION

### Reserve Design and Management

Ecologists have recently developed theories to guide the design of nature reserves so as to optimize their size, shape, and distribution (Diamond, 1975; Wilson and Willis, 1975; Simberloff and Abele, 1976; Game, 1980; Gilpin and Diamond, 1980; Higgs and Usher, 1980; Noss, 1983). Although most of the discussion of this subject has been carried on under the assumption that preserves are shaped exclusively by subtraction of existing natural habitat, adding habitat to reserves through restoration should also be considered. There are unquestionably preserves that would function more effectively as reservoirs for diversity if they were larger, for example, or if they were shaped differently, or if the populations within them were linked to others by habitat corridors. Such adjustments in preserve design frequently cannot be made simply by preserving existing communities, but they may be possible to achieve through restoration (Noss, 1985b), as has been suggested for a tropical dry forest preserve in Costa Rica (Janzen, 1985).

In fact, restoration is being used in this way at least occasionally by conservation organizations. Wilson and Willis (1975) even went beyond this to argue for programs of "planned biotic enrichment," envisioning the creation of new communities, the augmentation of diversity in existing ones, and creation of communities to provide habitat for specific "orphan" species on the brink of extinction.

### Responding to Changes in Climate

Most climatologists now believe it highly likely that the next century will see anthropogenic global warming on an ecologically significant scale (for example, NRC, 1983; Peters and Darling, 1985). Given the rapidity of the predicted changes and the fragmentation and isolation of many existing natural areas, many of these areas are likely to prove extremely vulnerable to these changes. Their survival may depend upon human ability to manage -- or even to move -- communities using techniques of community restoration.

### Right-of-way management

Between them, urban development, transportation corridors and public installations occupy an estimated 40 million hectares in the United States and are expected to increase to 49 million ha by 1990 (Klopatek et al, 1979). Generally regarded as an environmental "minus," these corridors offer opportunities for the reestablishment of certain kinds of native vegetation, and this approach to corridor maintenance is becoming more popular nationwide (e.g. Crabtree, 1984).

### Landscaping

Another motive for increasing the biodiversity of even some of the most heavily populated urban areas through restoration are the economic, environmental, and esthetic advantages of using assemblages of native plants in landscaping. This approach to "naturalized" landscaping has grown in popularity in recent years, especially in the prairie biome and in parts of the Southwest where the scarcity and high cost of water is an additional incentive for the development of landscaping methods that emphasize the use of water-conserving natives. In response to growing interest in this approach to landscaping a number of popular books have recently appeared (e.g., Diekelman and Schuster, 1982; Smyser, 1982).

### Agricultural Lands

Agriculture represents the single largest form of environmental disturbance in the U.S., and destuction of natural areas to create land for farming continues, even in areas of marginal agricultural value. As such lands decline in productivity and are abandoned, they represent major opportunites to increase diversity through ecological restoration (Klopatek, 1979). Indeed, there appears to be some interest in this approach to "recycling" farmland, especially in the arid southwest (Jackson, 1985).

### The Role of Ex Situ Conservation as a Source of Material for restoration

The relationship between *ex situ* conservation and restoration is a very close one, for one of the major purposes of *ex situ* conservation is to provide a source of propagules for reestablishment of wild populations. The degree to which *ex situ* populations will in fact be used in the future for either reintroduction of single species or for restoration of entire communities is hard to assess because there have been so few successful reintroductions (e.g., Konstanct and Mittermeier, 1982) and because the field of restoration ecology is so new.

Certainly attempts to reintroduce captive species will increase as more species become rare in the wild.

Organizations like botanical gardens and zoos are increasingly making _ex situ_ conservation of rare species a major goal and have begun to coordinate management of multiple _ex situ_ collections. Examples include the newly formed Center for Plant Conservation, which is organizing participating botanical gardens in an ambitious plan to maintain more than 3,000 rare U.S. plants in collections, and the Species Survival Plans for zoo animals, which attempt to manage all captive specimens of a given rare species as a single population.

Successful reintroductions of animals from captive populations are few (Botkin, 1977; Konstant and Mittermeier, 1982). Examples include the Arabian oryx; possibly the nene, or Hawaiian goose; the golden lion tamarin, which is a very recent effort; and future plans to reintroduce Przwalski's horse in Mongolia (Kleiman et al., 1985; P. Parker, 1985).

There have also been few plant reintroductions from _ex situ_ collections. One example is Knowlton's cactus (_Pediocactus knowltonii_), which is being cultivated by the Transition Zone Horticultural Institute in Flagstaff and which has also been reintroduced from cuttings by the U.S. Fish and Wildlife Service (McMahan, 1985). Some plant reintroductions, like that of the St. Helena redwood (_Trachetropis erythroxylon_), have failed after repeated efforts (Cronk, 1983), suggesting that _ex situ_ conservation, even when coupled with reintroduction, is not a certain method for ensuring long-term species survival.

In addition to programs designed to aid rare plant species, common plants may also be propagated by private and government nurseries specifically to provide material for restoration of plant communities, although in may cases the nurseries do not serve as true ex situ sources, particularly for the less commonly used plants, but rather gather seeds or vegetative materials from the wild (Jones, 1985; Kollar, 1985). This is especially true for relatively new types of restoration where horticultural techniques have not been worked out.

## KEY LEGISLATION

Legislation pertaining to restoration falls into three major categories. The legislation may directly provide for revegetation after a disturbance, as in the Surface Mine Control and Reclamation Act (SMCRA) and the National Forest Management Act. Alternately, it may require a reduction in environmental pollutants and therefore provide for natural recovery of stressed systems, as in the Clean Water and Clean Air Acts. Or it may be directed at preservation of native species, which could then be used in restoration efforts, as in the Endangered Species Act. The following is a brief discussion of current and potential legislation to encourage restoration oriented toward biological diversity.

The 1977 Surface Mine Control and Reclamation Act is currently the key legislation providing for revegetation following a disturbance. Sec. 515, para. (b)(19) states that mine operators shall "establish... a diverse, effective, permanent vegetative cover of the same seasonal variety native to the area of land to be affected..." However, this is interpreted by the Office of Surface Mining (30 CFR Part 816) as requiring reclamation, rather than restoration in our sense. The interpretation of the term "diverse" is still a point of controversy, but most state regulatory agencies, which administer

SMCRA, allow species mixtures of five to ten species where a stand of native species is to be established. The composition of the mixture may be varied even within one mined area according to variation in soil type, moisture, and exposure. Information, based on past experience, about which mixes of plants are most appropriate for the rehabilitation of particular mined lands may be provided by a variety of agencies, including the Bureau of Land Management, the Forest Service, and the Soil Conservation Service.

Sec. 515 para (b)(2) of SMCRA states that the mine operation shall "restore the land affected to a condition capable of supporting the uses which it was capable of supporting prior to any mining, or higher or better uses..." The phrase "higher or better uses," however is generally defined in economic terms, or in terms of social amenities like parklands. In the arid west, where the "higher" uses may be grazing or wildlife habitat, the planting of native species makes economic sense. In moister areas, however, like the eastern United States, "higher" economic uses may be, for example, plantation timber production, and restoration is very infrequent.

The only other major legislation that requires revegetation is the National Forest Management Act of 1976 under which reforestation is required on all forested lands in the National Forest System that do not regenerate naturally after timber harvesting. The lands are to "be examined after the first and third growing seasons... Any lands not certified as satisfactory shall be scheduled for prompt treatment." (Sec. 4 para. (d)(1)). However, "treatment" typically involves planting monocultures of tree species, which are unlikely to develop the diverse understory of the pre-logging forest. This act preserves the soil and the commercial productivity of the forests, but it is directed at biodiversity only in the sense that it advocates multiple use management, i.e., setting aside certain communities for uses other than timber harvesting.

Some final legislation that pertains directly to restoration is in the recently passed Farm Bill. The conservation reserve provision of this bill "authorizes USDA to contract with farmers to remove 40 million acres of erodible land from row crop production... The retired acres would be planted to grasses, legumes, and trees to reduce erosion and enhance wildlife" (Williamson, 1985). This bill will go a long way toward increasing biodiversity on marginal farmland, especially if native species were favored in the reseeding efforts.

A second series of legislative acts that pertain to restoration are aimed at reducing environmental pollutants. Control of sites where toxic substances have been deposited is vested in both The Superfund (Comprehensive Environmental Response, Compensation, and Liability Act of 1980) and the Resource Conservation and Recovery Act of 1976. Neither of these acts requires revegetation with native species, although either might be ammended to do so. An example of restoration which could be undertaken on a RCRA site would be, for instance, conversion of a closed sanitary landfill into a natural area.

The Clean Water, or Federal Water Pollution Control Act, has the goal of restoring and maintaining "the chemical, physical, and biological integrity of the Nation's waters" (Sec. 101 para. (a)). The Act deals primarily with the control of pollutants and maintenance of water quality. The Clean Water Act has promoted much natural recolonization of aquatic systems by native animals and plants. Legislative protection could be extended to enable the restoration of riparian and other watershed vegetation, thus reducing sedimentation from runoff water and providing microhabitats along the water's edge (e.g., shade, fallen logs).

The Clean Air Act is directed at public health and welfare but also has provisions to preserve the air quality in natural areas (e.g., national parks). Air cleanup may permit natural or assisted recolonization of previously toxic areas by sensitive native species.

The third type of legislation which affects restoration is legislation requiring environmental preservation. These laws protect communities that can serve as reference areas by which to judge the success of restoration efforts. The Endangered Species Act of 1973 is the most comprehensive of these, as it mandates conservation of endangered species as well as their habitats. Other legislation which is concerned with preservation includes the Wilderness Act of 1964, the Wild and Scenic Rivers Act of 1968, and portions of several other laws.

Several preliminary suggestions for legislative changes follow: The Surface Mine Reclamation Act could be strengthened so that state agencies would have to require a larger number of species in revegetation mixtures, although it must be recognized that complete restoration of all species is currently not economically feasible, nor even technically possible for all vegetation types. The new conservation provisions in the Farm Bill could also be strengthened by including legislation on restoration of vegetation in riparian areas. The National Environmental Policy Act (NEPA), which currently mandates consideration of the "quality" of existing systems, might be changed by adding the word "quantity," so that loss of area devoted to a particular community type might be made up through restoration. Additionally, the charge of the Council on Environmental Quality might be changed so that it also considers restoration in making environmental recommendations. Finally, specific legislation may be needed to restore those lands which are identified in this report as being the most depleted.

**Who is doing Restoration?**

We know of no systematic attempt to answer this important question, a fact that in itself underlines one of the key points we would like to convey in this report: While restoration technologies may have developed considerably during the past decade or so, there is still little tendency, either by environmental organizations or by the general public, to think about restoration as a discipline in its own right. Thus, most writing that deals with restoration is scattered through the literature of a dozen or so disciplines or appears in special publications narrowly focused on a particular problem or community tupe. This being the case, all we can do here is to make a number of generalizations that are based on our impressions of the field and that we feel are reasonably accurate.

First, while much reclamation is being carried out by industries in compliance with regulations, as one moves closer to the restoration end of the reclamation spectrum, one finds more and more of the work being done by small organizations and individuals. The individuals are often volunteers motivated by esthetic and environmental interests. This, in fact, is a theme of Restoring the Earth, a recent, popular account of restoration efforts in the U.S. (Berger, 1985).

Second, even among organizations which are primarily concerned with conservation and which do restoration rather than reclamation, restoration is often a secondary concern. It may be seen as an emergency measure for degraded areas, a measure to be discontinued once a community is able to sustain itself. Organizations which may engage in restoration as a matter of secondary importance include The Nature Conservancy, the Audubon Society, the National Park Service, and the Isaac Walton League.

Another group of organizations with a long history of work falling toward the reclamation end of the reclamation-restoration spectrum has a primarily economic orientation. This group includes the Bureau of Land Management, the U.S. Forest Service, and the Soil Conservation Service. Though the activities of these entities may be economically oriented, like reforestation efforts by the U.S. Forest Service, their projects often come close to restoration.

In addition to the preceding public and nonprofit organizations, the private sector plays an important role, often in the form of consulting companies who contract to do the actual restoration required by law of, for example, developers. Additional restoration work is done by landscaping firms, and a scattering of specialty nurseries has developed in response to the resulting demand for native plants. A few of these (notably NPI, in Salt Lake City) are large concerns; most are small, struggling business.

University programs are also generally oriented toward a reclamation rather than restoration approach. Of the several dozen reclamation-oriented programs established at U.S. colleges and universities during the past decade, most if not all are reclamation-oriented (Sagal and Jordan, 1986). Although some courses may deal restoration, we know of none devoted specifically to the subject. However, some evidence of change is visible: restoration has been identified as a focus for research at the Cary Arboretum in New York, and a Center for Restoration Ecology has been created at the University of Wisconsin-Madison.

## LIMITING FACTORS

One of the most important factors limiting the application of restoration to the conservation of biodiversity is that, in spite of the fact that technical advances have been rapid, there is still little widespread appreciation of the potential contribution of restoration, even among the environmentally oriented organizations and agencies concerned with managing degraded lands. To some extent this is because of the belief that restoration's use is most appropriate to small scale projects and to lands that have been severely degraded. Contrarily, as we have discussed, restoration can be used on a fairly large scale, the restoration of the Kissimmee River in Florida being a good example, and is valuable for upgrading less severely damaged lands. Lack of attention may also reflect the fact that restoration falls in a kind of economic and psychological no-man's land, appealing neither to agricultural interests nor to environmentalists concerned with the preservation of pristine natural areas. Thus, a major challenge for conservationists will be convincing the government agencies that manage land and fund research to see restoration as a better alternative than reclamation for a wider variety of situations.

An additional reason for restoration's low profile may be because the discipline has not yet fully crystalized. As we have said, the literature is scattered and there is an urgent need for better communication among professionals working on restoration projects, both practical and theoretical. Crossley (1984) surveyed communications in this area, and found them "sorely inadequate". In general, so far as we know there exists no central source of information about restoration and no organization committed to articulating its implications or disseminating information about it. Indeed this fact has complicated recent efforts by the Department of the Interior to prepare guidelines for use of Superfund monies for environmental cleanup (Anonymous, 1983).

An additional need is for greater support of restoration research, as compared to practical restoration attempts, to elucidate fundamental, generalizable principles and to foster a closer link to ecology per se. To encourage this development, the University of Wisconsin-Madison Arboretum has recently introduced the term "restoration ecology" to refer to the restoration of ecological communities as a technique for basic ecological research (Aber and Jordan, 1985; Jordan, Gilpin and Aber, 1986).

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